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Symposium on
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Planning,
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disturbed lands

PLANNING, REHABILITATION AND TREATMENT OF DISTURBED LANDS

Sixth Billings Symposium
March 21-27, 1993

Volume II: Study of a Superfund Site - Butte/Anaconda and Silver Bow Creek
Woody Plant Species
Hydrology
Wetlands/AMD Development
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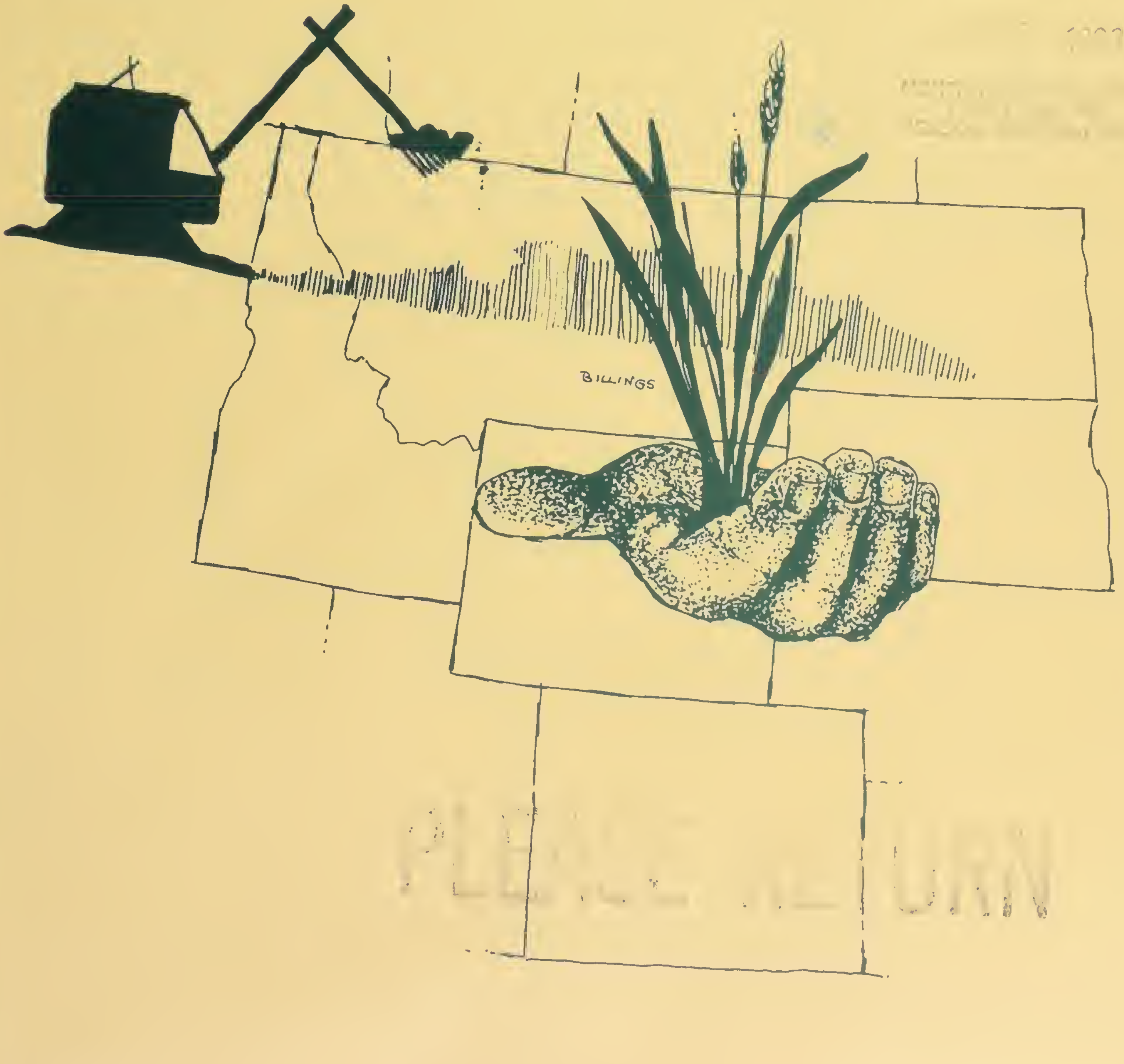

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Frank F. Munshower
Co-Chairman

Planning, Rehabilitation and Treatment of Disturbed Lands

Sixth Billings Symposium
March 21-27, 1993

Billings Plaza Holiday Inn, Billings, Montana

Volume II

CO-CHAIRMEN

Frank F. Munshower
Reclamation Research Unit
Montana State University
106 Linfield Hall
Bozeman, MT 59717-0290

Scott E. Fisher, Jr.
Office of Surface Mining
Western Technical Center
1020 15th St., Brooks Towers
Denver, CO 80202

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VOLUME II

TABLE OF CONTENTS

Page

Study of a Superfund Site - Butte/Anaconda and Silver Bow Creek

Streambank Tailings Revegetation 1
D.R. Neuman, F.F. Munshower, D.J. Dollhopf, S.R. Jennings, W.M. Schafer and J.D. Goering

Modeling the Fate and Transport of Metals in Surface Water at the Silver Bow Creek CERCLA Site 17
W.M. Schafer, J.D. Goering, T.R. Grady, E. Spotts and D.R. Neuman

Anaconda Revegetation Treatability Study 32
R.J. Montgomery

Two Methods of Reclaiming Stream Ecosystems Impacted by Mine Tailings 42
C.T. Stilwell

The Clark Fork Basin, Montana Regional Concepts for Mega-Site Remediation 45
M. Bishop

Woody Plant Species

Can We Improve Shrub Seedling Vigor By Managing Seed Imbibition? 47
D.T. Booth

Use of Prescribed Burning in Regenerating Woody Draws 59
C.H. Sieg

Mycorrhizae and Mined Land Restoration: Current Status and Future Directions 60
H. Atthowe

Utilization of Site-Adapted Containerized Plant Material for Successful Woody Plant Restoration 72
V. Pfannenstiel, L. Ballek and J. Krueger

TABLE OF CONTENTS (Continued)

Page

Woody Plant Species (Cont.)

Mature Shrub Transplant Success on Reclaimed Mine Spoils in Northwest Colorado 82
W.S. Harding, Jr.

Hydrology

Remediation of Water Quality Problems Associated With Drainwater From Department of the Interior Irrigation Projects 98
R.A. Engberg and A.J. Cappellucci

Computer Simulations of the Influence of Suspended Iron-Rich Particulates on Trace-Metal Removal From Mine-Drainage Waters 107
K.S. Smith, W.H. Ficklin, G.S. Plumlee and A.L. Meier

Cover/Soil Effects on Reclaimed Mineland Erosion 116
S.A. Schroeder

Erosion and Sediment Control Product Selection for Drastically Disturbed Lands 124
W. Agnew and M.S. Theisen

Unexpected Impacts to Unmined Aquifers Near Coal Mines 134
J. Reiten and J. Wheaton

Bentonite as a Hole Abandonment and Annular Seal Material 143
S. Regele, D. Clark, B. Bohman, J. Wheaton and J. Lyford

A Bioengineering Technique for Stream Channel Reconstruction 151
D. Miller and J.A. Lovell

Wetlands/AMD Development

Raising the Water Table to Create a Wetland 158
S. McIntosh

Evaluation of Organic Substrates for Use in Wetlands Constructed to Treat Acid Mine Drainage 170
E. Spotts, T.S. Mitchell, C.T. Hoschouer and W.M. Schafer

TABLE OF CONTENTS (Continued)

Page

Wetlands/AMD Development (Cont.)

Wetland Reclamation at a Surface Coal Mine in Montana 183
B.C. Waage

Nutrient Removal and Plant Root Responses to Applications of Livestock Effluent Using Constructed Wetlands 184
B.R. Romig and R.W. Tierney

Cyanide

Reclamation-Mining of Sulfide-Rich Mine Dumps Using Heap-Leach Techniques: Geochemical Implications of Cyanide Leach Column Tests 186
W.H. Ficklin, G.S. Plumlee, J.H. Bullock, Jr., M. Stewart and B. Phillips

Field Evaluation of Land Application Performance: Metals Removal From Barren Leach Solution 195
S.C. Smith, T.J. Hudson and W.M. Schafer

Residual Cyanide Distribution in a Neutralized Heap Leach Pad 208
J.G. Poell and F.F. Munshower

General Reclamation

Bird Mortalities Associated With Mining and Oilfield Waste Pits in Wyoming 217
P. Ramirez, Jr. and B.J. Esmoil

SMCRA-Inception, Maturation, and What? 222
L.L. Byrd

Addendum

Integrated Suitability Criteria for Soil and Overburden Materials 226
S.E. Fisher, Jr. and T.H. Brown

Influence of Rhizosphere Microorganisms on Uptake of Selenium by Two-Grooved and Tine-Leafed Milkvetches 227
S.E. Williams

TABLE OF CONTENTS (Continued)

Page

General Session and Poster Sessions

Revegetation Success and Bond Release: Where Do We Stand? 234
V.J. Bryan and R.F. Price

Abandoned Bentonite Mine Spoil Reclamation - 12 Years of Research 242
G.E. Schuman, E.J. DePuit, E.M. Taylor, Jr. and J.L. Meining

The City of Anaconda Erosion Control and Stabilization of "C" Hill 246
L. Holzworth, J. Schaefer, G. Green and T. Wiersum

Using Organic Matter to Select Optimum Topsoil Salvage Depths: A Strategy to Improve Topsoil Quality and Reduce Salvage Depths 253
K.D. Coppinger, P.R. Ogle and D. McGirr

The Adaptability of Selected Tree and Shrub Species for Surface Mine Reclamation 263
M.J. Knudson, R.J. Haas, D.A. Tober and E.T. Jacobson

The Statistical Design and Evaluation of Revegetation Data for Determining Revegetation Success 264
R.C. Postle

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

STREAMBANK TAILINGS REVEGETATION
SILVER BOW CREEK, MONTANA

D.R. Neuman[1], F.F. Munshower[1], D.J. Dollhopf[1], S.R. Jennings[1], W.M. Schafer[2] and J.D. Goering[2]

ABSTRACT

The evaluation of *in situ* chemical immobilization of contaminants in mine wastes/soils followed by revegetation as a potential remedial alternative for cleanup of a Superfund site is described. This six-year treatability study evaluated the use of different amendments, various incorporation techniques, and selected plant species in both a laboratory/greenhouse setting and at five waste/contaminated soil sites located along Silver Bow Creek between Butte and Opportunity, Montana.

Amendments (chiefly $Ca(OH)_2$ and $CaCO_3$) found in the laboratory to be most effective in controlling pH of these acid producing metalliferous wastes and reducing soluble contaminants were added to wastes/soils in replicated greenhouse trials. Amendments and greenhouse selected plant species were combined into a replicated field trial of different amendment incorporation techniques. Components of the soil chemistry, soil hydrology, and vegetation response were measured and differences among treatments were evaluated.

The data support coversoils as most effective. However, coversoils are not available in quantities adequate for the rehabilitation of all of the disturbances. The pressure injection of lime slurry into the wastes did not provide an adequate rootzone for long-term plant growth. Agricultural tilling provided only 15 cm of adequate rootzone materials. A plow capable of mixing waste to a depth of 122 cm provided an amended rootzone that varied from 30 to 60 cm deep. Plant cover and production were greater on deep plowed plots than on plots prepared by the other two incorporation techniques. Deeper root penetration was also found in the deep plowed plots than in the plots treated by the agricultural plow or the injection technique. Surface runoff was markedly reduced and surface water quality improved by all treatments.

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[1] Reclamation Research Unit, Montana State University, 106 Linfield Hall, Bozeman, MT 59717-0290.

[2] Schafer and Associates, 865 Technology Blvd., Bozeman, MT 59715

INTRODUCTION

Silver Bow Creek originates north of Butte, Montana, and is a major tributary to the upper Clark Fork River. Mill tailings and other mining wastes in and near the creek contaminate downstream areas with potentially toxic elements: arsenic, cadmium, copper, lead, iron, and zinc. These elements and others were present in the mine ore and remain as by-products of the milling and smelting processes. Silver Bow Creek and contiguous portions of the upper Clark Fork River were listed as a Superfund site in 1983 by the United States Environmental Protection Agency (EPA) under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA). The site extends from Butte to Milltown, Montana, some 225 km (140 river miles).

Remediation of these wastes is the focus of RI/FS activities on the Streambank Tailings Operable Unit of the Silver Bow Creek investigation. Streambank tailings are eroded by wind and water. Sulfide oxidation in the mine wastes increases acidity which increases the mobility of most metals. As water percolates through or flows across the barren streambank mine wastes, metals are leached into ground and surface waters. Water evaporating from the land surface leaves a residue of metal-enriched salts which can degrade air quality.

TREATABILITY STUDY DESCRIPTION

Remediation of stream deposited mining wastes in the Silver Bow Creek channel and floodplain using low-cost, *in situ*, innovative technologies was the major goal of the Streambank Tailings and Revegetation Studies (STARS). The approach conceptualized by the project team (Reclamation Research Unit and Schafer and Associates) was a treatability study involving the chemical neutralization and fixation of contaminants by mixing amendments into the waste materials followed by revegetation using acid, metal, and drought tolerant species of vegetation. The soil amendments would neutralize acidity of the wastes, reduce mobility of metals, and reduce their toxicity. The vegetation would reduce transient dust, water erosion and surface water runoff. The vegetation would also harvest water moving through the rootzone.

The STARS Treatability Study consisted of three phases:

- Phase I: Bench-scale, soil-column and greenhouse investigations

Objectives of Phase I were to: 1) test a variety of chemical amendments for their effectiveness in neutralizing tailings acidity and in reducing contaminant mobility; 2) evaluate acid, metal and drought-tolerant plant species for their abilities to thrive in the amended wastes; and 3) develop a ranking system for classifying tailings based on easily identifiable physical and chemical traits. Results of Phase I are described by Reclamation Research Unit et al. (1989).

- Phase II: Pilot-scale treatability study for field implementation

Objectives of Phase II were to implement in the field amendments and vegetation that were found to be most effective during Phase I. One hundred field plots were established at five different tailings sites along Silver Bow Creek. The plots were treated with amendments (using different mixing methods), fertilized, and seeded with appropriate vegetation. Details of Phase II are described by Schafer and Associates et al. (1989a).

- Phase III: Field monitoring program

The primary purpose of Phase III monitoring was to provide the Feasibility Study team with appropriate data on the effectiveness of the STARS technology to abate contaminant movement and reduce toxicity of the waste materials. This was accomplished by: 1) measuring components of soil hydrology, soil chemistry, and plant performance in each study plot; 2) comparing these data among the treatments, at each site, to determine their relative effectiveness; and 3) use modeling to predict long-term effectiveness of the treatments. Details of Phase III are described by Schafer and Associates and Reclamation Research Unit (1993).

Phase I: Bench-Scale Neutralization Studies: Laboratory and Greenhouse Investigations

Existing data concerning tailings and contaminated soils along Silver Bow Creek were reviewed by the project team. This information guided the selection of 35 contaminated sites within the Silver Bow Creek ecosystems. These sites were representative of the wastes along the 40 km of this stream. From each of these 35 sites, a 225 kg bulk soil or waste sample was collected. These samples were dried, sieved, mixed, and subsampled for analysis of key chemical and physical characteristics. Six primary "kinds" or groups of waste types were statistically identified on the basis of this chemical and physical characterization. These six types of wastes/contaminated soils were representative of the full spectrum of contamination in the Silver Bow Creek area. A representative example of each group or kind of waste was selected for further analyses in the laboratory and greenhouse.

Concurrent with the selection of mine wastes or tailings for further study both traditional and innovative amendment materials were evaluated for possible use in the greenhouse and field studies. Candidate amendments were evaluated in terms of their ability to raise waste pH, reduce the solubility of metals, availability of the amendment, and cost. Materials evaluated included calcium carbonate ($CaCO_3$) (agricultural lime), cement kiln dust, calcium chloride ($CaCl_2$), calcium hydroxide ($Ca(OH)_2$), calcium oxide (CaO), power plant flyash, sugarbeet waste, gypsum, zeolite, triple superphosphate, fluoro-phospho silicate, phosphogypsum ferric sulfate ($FeSO_4$), phosphate ore, and various organic amendments. The chemical and physical characteristics of these amendment materials were determined. The amendments were mixed with the tailings or waste materials in specially designed reaction flasks (highly modified Tempe Cells) and their ameliorative properties in controlled and replicated laboratory column studies determined (Reclamation Research Unit et al. 1989). Quantities of the amendments necessary to neutralize various wastes were measured.

Effluents from the Tempe Cells were evaluated for control of pH and metals by the various amendments. The leached and amended tailings or wastes were also analyzed. Results revealed that the amendments that most significantly reduced acidity were also most effective in limiting the concentration of soluble metals. Combinations of $CaCO_3$, $Ca(OH)_2$, phosphogypsum, Fe_2SO_4, $CaCl_2$, and triple superphosphate performed best and were subsequently tested in greenhouse plant growth trials.

An equation for the calculation of the quantity of lime was developed:

$$t\ CaCO_3/1000\ t\ Soil = 1.25\ (A + B)$$

where t = tons
A = Potential Acidity
B = Active Acidity
Potential Acidity = 31.25 (% HNO_3 S + Residual S) + 23.44 (% HCl S)

where S = sulfur content extracted from or remaining in the sample
Active Acidity = SMP Buffer Test

The use of calcium carbonate alone did not raise the pH fast enough or high enough to permit seeding within a reasonable amount of time. It was found that portioning the liming rate into 60% calcium carbonate and 40% calcium oxide rapidly raised the pH and reduced metal toxicity. All amendments were composed of these two calcium sources in these proportions.

Plant species used in the greenhouse trials were selected on the basis of their ability to tolerate acidic soil conditions, nutrient deficiencies and the climatic conditions at the Silver Bow Creek site (Reclamation Research Unit et al. 1989). In addition, some species were selected for site specific characteristics such as elevated salinity or textural extremes. Finally, all species were screened for commercial availability. Seventeen species and two cultivars of two species were grown in amended tailings in the greenhouse.

At the termination of the greenhouse growth period, the plants were evaluated for vigor, height, color, root growth, and production (Reclamation Research Unit et al. 1989). Plant tissues were also analyzed for metal levels. Livestock and wildlife toxicities from elevated metal levels were thought to present a concern if the area is revegetated. In the plant tissues analyzed, concentrations of boron, calcium, and lead were comparable to tissue levels in nonpolluted areas of Montana. Cadmium, copper, manganese, and zinc were elevated compared to background levels in Montana but these concentrations would not be toxic to any grazers and they were not phytotoxic. Aluminum and arsenic levels were higher than most background values in the literature, but they would not be toxic to animals. The higher aluminum values were high enough to indicate some plant growth inhibition. Metal loading in and on plants grown in the field were much higher. These data are discussed later in this paper. Major results of STARS Phase I are exhibited in Table 1.

Phase II: Pilot-Scale Treatability Studies - Field Implementation

The implementation of several amendment incorporation techniques (see Table 1) was described in STARS Phase II. Blocks of five or six treatments were implemented at five different locations within the Operable Unit. Waste material at each site represented the different types of tailings found along Silver Bow Creek (see Table 1). The treatment blocks were replicated four times at each site to provide for statistical comparisons of the response variables. The Manganese Stockpile (STARS Site 2) was located within the Butte Reduction Works, east of the present day Butte Sewage Treatment Plant. STARS Site 7 was located adjacent to Silver Bow Creek near the town of Rocker. The large expanse of flood deposited wastes near the town of Ramsay was the location of STARS Site 27 (Ramsay Flats). The fourth STARS site (27) was situated north of Fairmont near the "Yellow Ditch", while the last site (33) was located adjacent to Silver Bow Creek near the town of Opportunity.

The soil amendments selected from STARS Phase I were incorporated into the waste materials using five techniques at each field site in Phase II. Treatment 1 involved standard agricultural tillage to a depth of 15-20 cm (6-8 inches). A small tractor with a chisel plow,

Table 1. Type and characteristics of streambank wastes, effective amendments, and successful vegetation found in Stars Phase I.

Tailings/Waste Type	General Characteristics (Range or Mean)	Effective Laboratory Amendments and Rates (tons/1000 tons waste)		Successful Greenhouse Vegetation	Representative Field Site
In-Place Tailings (Man-made ponds)	pH 2.7 - 4.2 EC 4.9 - 6.7 mS/m As 1000 mg/kg Cu 4500 mg/kg Zn 10000 mg/kg	$Ca(OH)_2$ $CaCO_3$ $Fe_2(SO_4)_3 \cdot 7\ H_2O$	13.2 50.5 2.0	Altai wildrye (*Elymus angustus* Prairieland) Pubescent wheatgrass (*Agropyron trichophorum)* Tall wheatgrass (*Agropyron elongatum* Orbit)	Manganese Stockpile STARS Field Site 2
Class 1 (Fluvially deposited)	pH 3.0 - 4.0 EC 2.0 - 5.0 mS/m Coarse 50 - 85% Silt 10 - 30% As 250 mg/kg Cu 1200 mg/kg Zn 1200 mg/kg	$CaCO_3$ Phosphogypsum	56.1 5.0	Western wheatgrass (*Agropyron smithii* Rosana) Basin wildrye (*Elymus cinereus* Magnar) Pubescent wheatgrass (*Agropyron trichophorum* Luna) Altai wildrye (*Elymus angustus* Prairieland)	Near Opportunity STARS Field Site 33
Class 2 (Fluvially deposited)	pH 3.5 - 4.5 EC 0.5 - 2.5 mS/m Coarse > 85% Silt < 10% As 80 mg/kg Cu 250 mg/kg Zn 1200 mg/kg	$CaCO_3$ Phosphogypsum	39.9 5.0	Canadian bluegrass (*Poa compressa* Common) Mammoth wildrye (*Elymus giganteus* Volga) Tufted hairgrass (*Deschampsia caespitosa)*	Near Rocker STARS Field Site 7
Class 3 (Fluvially deposited)	pH 3.5 - 5.5 EC 4.0 - 12.0 mS/m Coarse < 50% Silt > 30% As 1000 mg/kg Cu 2500 mg/kg Zn 4000 mg/kg	$Ca(OH)_2$ $CaCO_3$ $Fe_2(SO_4)_3 \cdot 7\ H_2O$	17.7 35.7 2.0	Tall wheatgrass (*Agropyron elongatum* Orbit) Saltgrass (*Distichlis spicata)* Basin wildrye (*Elymus cinereus* Magnar) Western wheatgrass (*Agropyron smithii* Rosana)	Ramsay Flats STARS Field Site 21
Flood Affected Lands	pH 4.4 - 5.3 EC 0.3 - 5.0 mS/m As 250 mg/kg Cu 200 mg/kg Zn 20 mg/kg	$Ca(OH)_2$ $CaCO_3$ TSP	3.3 16.6 1.1	Basin wildrye (*Elymus cinereus* Magnar) Crested wheatgrass (*Agropyron cristatum* Ephraim)	Near Fairmont STARS Field Site 27

moldboard plow, and rototill was used. Treatment 2 involved deep (1.2 m, 4 ft) mechanical plowing of amendments into tailings with a plow pulled by a D8-H Caterpillar. For Treatment 3, a aqueous slurry of lime was pressure injected 1.2 m (4 ft) into the waste materials. Treatment 4 involved incorporating amendments by standard agricultural tillage, followed by constructing a coversoil wedge ranging from 0 to 46 cm (0-18 inches) of material. Treatment 5 was a control which received no amendments. After all amendments were incorporated, water was applied and rototilled into the plots to encourage the hydration of lime and reaction with the waste materials. Fertilizers, determined by site characteristics, were applied. The fertilizer was rototilled into the plots which prepared the surface for seeding.

Seeding recommendations for each of the five sites were based on the greenhouse evaluation of plant performance in Phase I. Two seed mixtures were recommended for each site. Seeds of recommended species for each mixture were weighed and placed in individual packages for each study plot. Sites 7, 21, 27 and 33 were seeded April 12-14, 1989. Site 2 was too wet to seed during this period and was seeded on May 19, 1989. A detailed description of the construction of the experimental plots and implementation of the treatments is provided in the Phase II Final Summary Report (Schafer & Associates, Reclamation Research Unit, and CH2M Hill, Inc. 1989).

Phase III: **Monitoring and Evaluating Field Plots**

The primary purpose of the STARS Phase III monitoring program was to measure the effectiveness of STARS treatments to abate contaminant movement and reduce the phytotoxicity of the tailings. This was accomplished by 1) measuring soil chemical, physical, and hydrologic properties, and evaluating plant performance at each site; and then 2) comparing data collected for each treatment to determine relative differences in measured properties. Treatment effectiveness will be inferred in this document from these data. Specific data collection needs were:

- measure soil chemical properties and identify differences between treatments that may effect contaminant solubility and bioavailability, or that may affect long-term treatment effectiveness;
- evaluate erosion (soil loss);
- measure dissolved and total metal concentrations in surface runoff;
- measure metal cation and anion concentrations in soil pore water;
- measure metal loadings in forage plants; and
- measure the survival and performance of seeded vegetation.

SOIL CHEMICAL MONITORING

Soil pH, EC levels, and concentrations of soluble metals were measured in saturated paste extract solutions prepared from amended tailings/soils collected from various depths within the profile. Profiles were collected from each experimental plot. Concentrations of bio-available metals were determined in extracts of ammonium acetate (pH 7.0), while bio-available arsenic levels were measured in Bray P-1 extracts of the amended tailings/soils. Surface materials (0-2.5 cm) from each plot were collected from each STARS experimental plot including the control plots. Surface materials outside, but adjacent to each replication, were also collected. These samples represent pre-treatment and the no action alternative. The collected samples were analyzed for total metal concentrations. The degree of mixing of amendments and wastes was evaluated in the final year of the study (1992) by spraying a pH indicator dye on the face of excavations made to a depth of 2.1 meters (7 feet) in several STARS plots. Zones of different pH were readily discernable and volumes of wastes material were described (see Schafer and Associates and Reclamation Research Unit 1993).

Figure 1 exhibits pH response to the STARS treatments at Site 2 (Manganese Stockpile) and Site 7 (Rocker). At Site 2, Manganese Stockpile (Figure 1, top), the pH values of the materials in the treated plots were much higher than the control plot wastes, especially for the upper depth increments. At a tailings depth of approximately 40 cm, wastes treated by agricultural tillage revealed a similar mean pH to the untreated materials. In contrast, wastes amended with lime using the two deep incorporation techniques exhibited elevated mean pH levels throughout the profile.

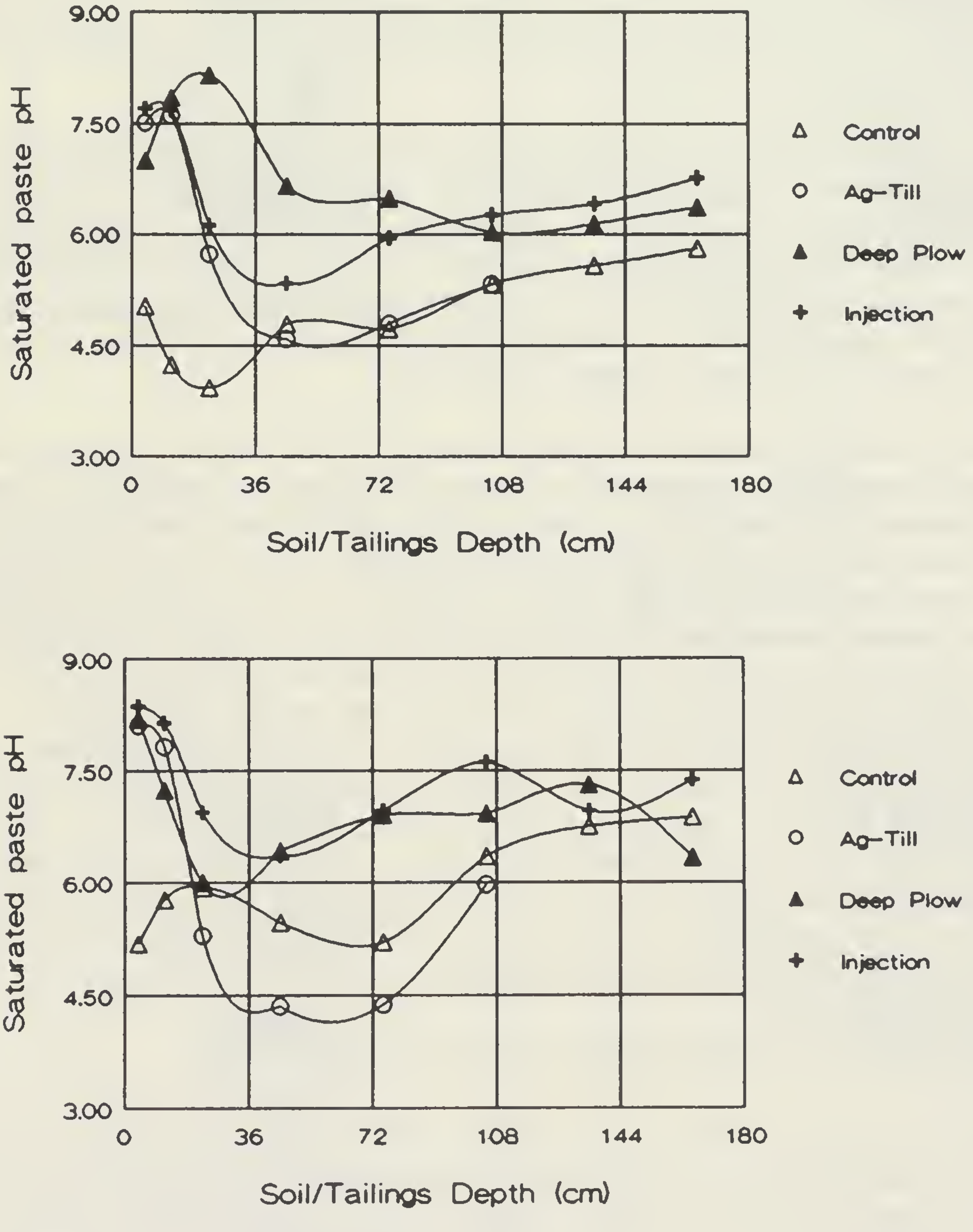

Figure 1. Saturated paste pH levels and tailings/soils depth for Sites 2 (top) and 7 (bottom).

At Site 7, Rocker (Figure 1, bottom), mean pH levels of the upper amended wastes layers (0-15 and 15-30 cm) were elevated compared to the mean value of the control plots. Low pH levels were encountered below the zone of incorporation for the agricultural tillage treatment. At the deepest profile depths, the pH of the control, deep plow and injected treatment plots were all similar. These depths were below the amendment incorporation zone and the relatively high pH values most probably indicate these materials were natural soils - not effected by the overlaying mine waste materials. Similar patterns of pH, depth of treatment, and depth to natural soils were exhibited at the other three STARS sites.

The impact of the STARS treatments and measured electrical conductivity of the amended wastes/soils indicated elevated levels compared to the control plot values in the upper profile depths. Below the zones of amendment incorporation, the measured EC levels of materials in the treated plots were similar to the wastes/soils in the control plots. Use of calcium carbonate, calcium hydroxide and the other amendments in the STARS treatments not only affected pH; but also changed the composition of the soluble salts. Metal contaminants (copper, zinc, cadmium, lead, etc.) were removed from the soil solution and were replaced by calcium. As a result, the pH of the amended wastes was ameliorated and the metal contaminants in the soil solution were replaced by a less phytotoxic cation - calcium.

Mean water soluble metal levels in the upper 15 cm of the wastes/soils are shown for four of the STARS sites in Figure 2. In general, concentrations of water soluble Cd, Cu and Zn were reduced by orders of magnitude compared to the control plot levels. Concentrations of Pb were slightly reduced, but levels of As were enhanced, especially at Sites 21 and 27. This increased As mobility is clearly exhibited in Figure 3 (top). Only slight (nonsignificant) increases in As levels were found in the lower profile depths. In contrast to the increased As availability, the decreased concentration of soluble Cu is displayed in Figure 3 (bottom).

Relationships between bioavailable concentrations of As, Cu, Cd, and Zn in the amended wastes/soils and levels found for the vegetation are exhibited in Figure 4. Ammonium acetate extracts of the wastes/soils were analyzed for levels of Cu, Cd, Zn and other metals, while the Bray P-1 extracts of the materials were analyzed for As concentrations. Pairs used in the statistical analysis were the depth integrated mean bioavailable concentrations for a specific experimental plot and the concentrations for the species-specific plants growing in that plot. Arsenic found in ryegrass plants was positively correlated to the measured bioavailable waste/soil As levels. The correlation ($r = 0.638$) was significant at the $p \leq 0.05$ level. The amount of As found in wheatgrass samples was negatively correlated (nonsignificant) to the extractable waste/soil concentrations (Figure 4). The reason(s) for the difference in correlation between the wheatgrass and the wildrye grasses is unknown. When the data for As were combined, the r value was -0.082 indicating no relationship between bioavailable As as measured in Bray P-1 extract of the amended wastes/soils and As found in the vegetation grown in the test plots. Relationships for the metals measured in the ammonium acetate extract and metals in the vegetation are also shown in Figure 4. All correlations were positive and statistically significant ($p \leq 0.05$).

VEGETATION MONITORING

Plant performance on the STARS plots was inferred from measurements of species areal cover, species production, plant rooting depth, and plant species metal levels. Percent areal cover, measured in all three years, indicated the relative effectiveness of the establishing vegetation to reduce leaching, surface runoff, and fugitive dust. Plant production (peak standing crop) indicated potential future land uses of the tailings once they are successfully reclaimed, assuming plant metal

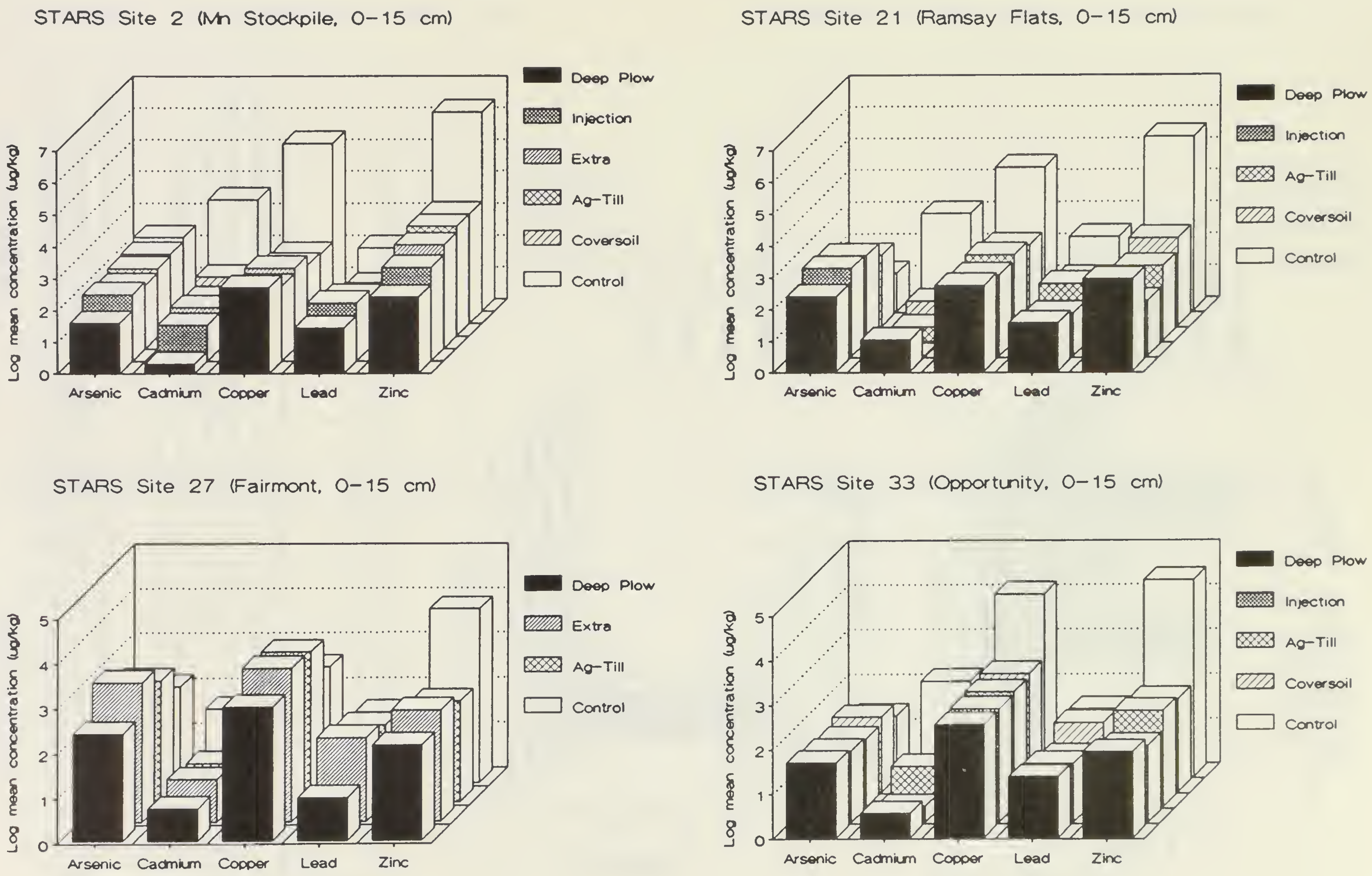

Figure 2. Mean water soluble soil (0-15 cm) metal and As levels for STARS Sites 2, 21, 27 and 33.

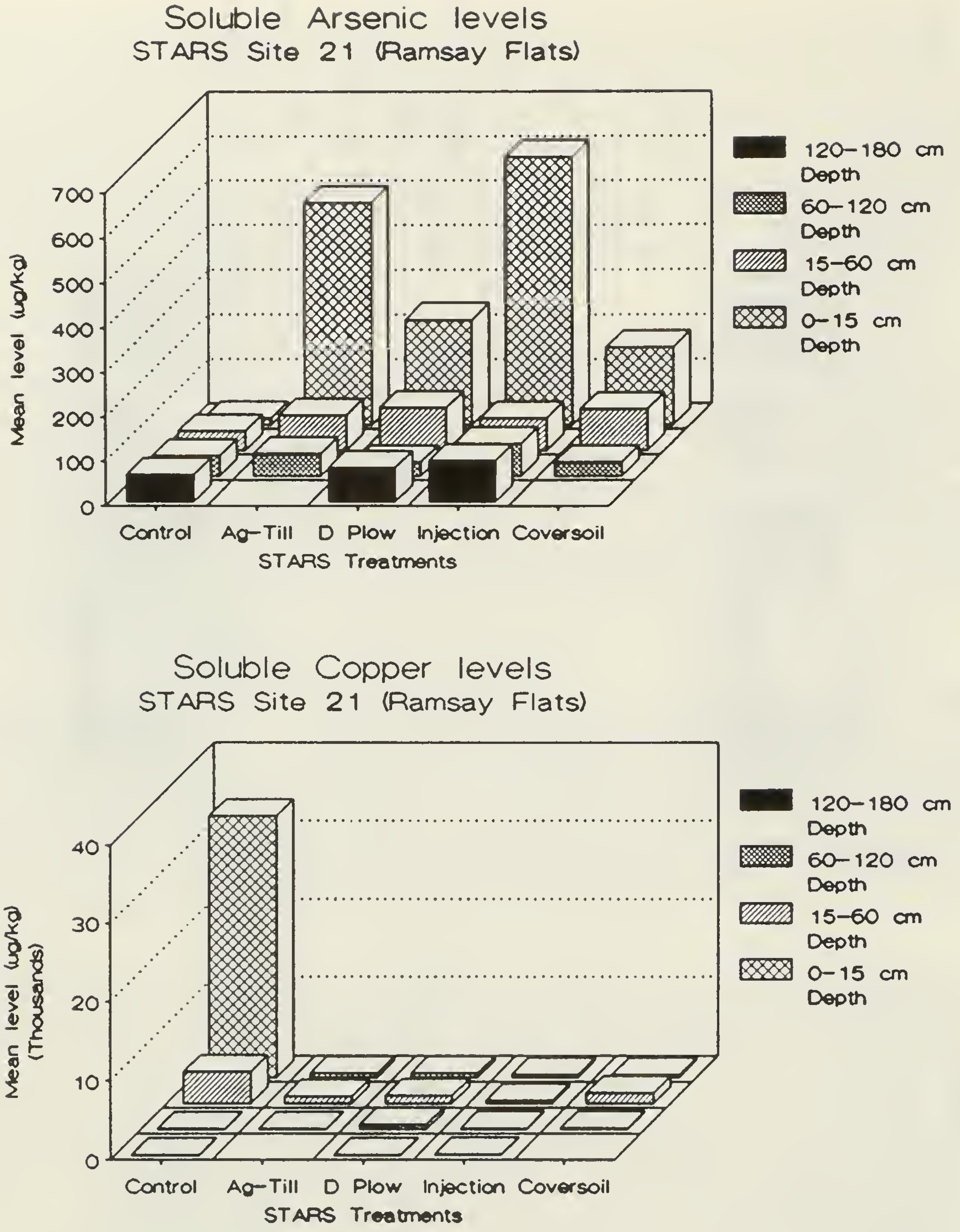

Figure 3. Mean water soluble soil As (top) and Cu (bottom) levels at STARS Site 21-Ramsay Flats arranged by profile depth and treatment.

levels are not toxic for grazing animals. Rooting depths indicated long-term plant community stability; plants with root systems limited to upper soil layers would be less drought tolerant than those that can penetrate deeper layers. When combined with soil metal levels, plant metal levels indicate whether plants are accumulating metals, possibly passing them into the food chain, or excluding them. If plants accumulate metals, there may be a redistribution of metals from subsurface to surface soil layers as plant materials decompose.

At almost every site, vegetation production and cover on the Control plots was significantly lower than the same parameters on the amended plots. When these two parameters were measured on the Coversoil Wedge they were usually comparable to these same parameters on the

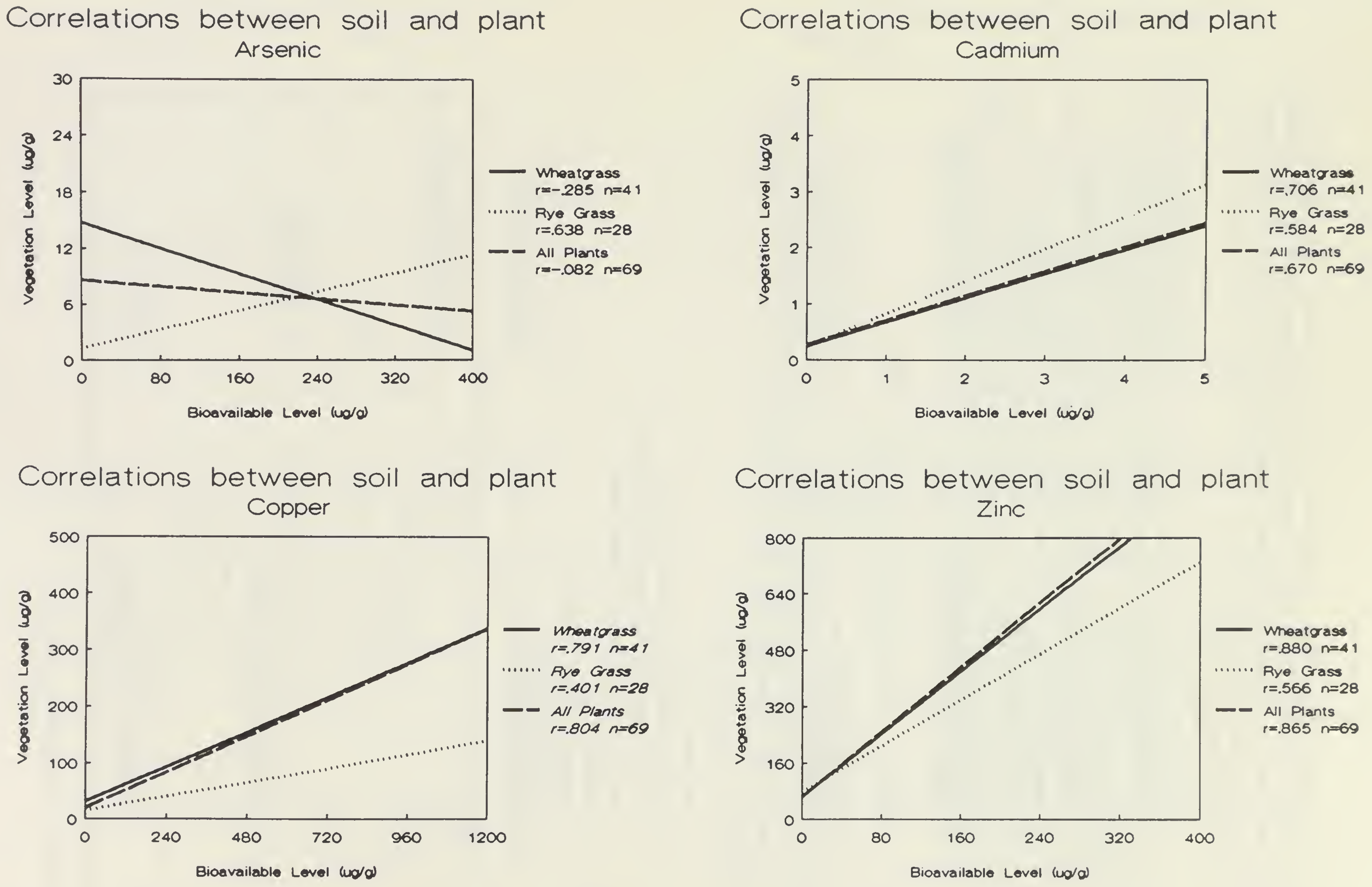

Figure 4. Relationship between soil and plants As, Cd, Cu and Zn levels.

Deep Plowed treatment. These trends are graphically exhibited in Figure 5. Plant metal levels were lower, however, on the topsoiled wedges. Because of the large variation in Control plots and metal levels, there were few significant differences among the STARS treatments at a site. When differences occurred among the STARS treatments, they were almost always between the deep plowed and the other STARS treatments.

To designate the plant species that performed well on each waste type, Importance Values were generated. This number provides an objective, data based measurement of the *importance* of each seeded species at each site (Table 2). These species were easy to establish and provided better cover and production than the other seeded species. Long-term survival of these species cannot be determined at this time. Metal levels in and on these species at Sites 7, 27, and 33 were generally within ranges recommended for livestock consumption. Cadmium was the notable exception. Concentrations of this element often exceeded recommended tolerances for livestock consumption. Metal and As concentrations were very high for dominant species at Site 21 (Table 3), and it is inferred that these high metal levels were due to dust and rainfall splash. Contaminants were on the plants rather than in the vegetation.

SOIL HYDROLOGY MONITORING

Within each experimental plot in one replication block at each site, the following soil hydrologic monitoring equipment was installed: neutron access tube, piezometer, and up to three pore water samplers (suction lysimeters). In addition, a climatic station was installed at STARS Site 21 (Ramsay Flats) and precipitation gauges were installed at the other sites. The following data were generated:

- soil water content;
- soil pore water pH, SC and Eh levels;
- soil pore water metal, cation and anion levels;
- measurements of depth to groundwater;
- climate; and
- simulated rainfall/runoff tests.

Table 2. Species exhibiting highest Importance Values.

Site	Species	Overall Rating[1]
7	Mammoth wildrye Altai wildrye Thickspike wheatgrass	109.2 103.2 36.4
21	Tall wheatgrass Intermediate wheatgrass Russian wildrye	184.9 125.3 59.5
27	Desert wheatgrass Crested wheatgrass Russian wildrye	144.9 101.3 72.8
33	Altai wildrye Basin wildrye	103.7 51.8

[1] Grand mean of Importance Values.

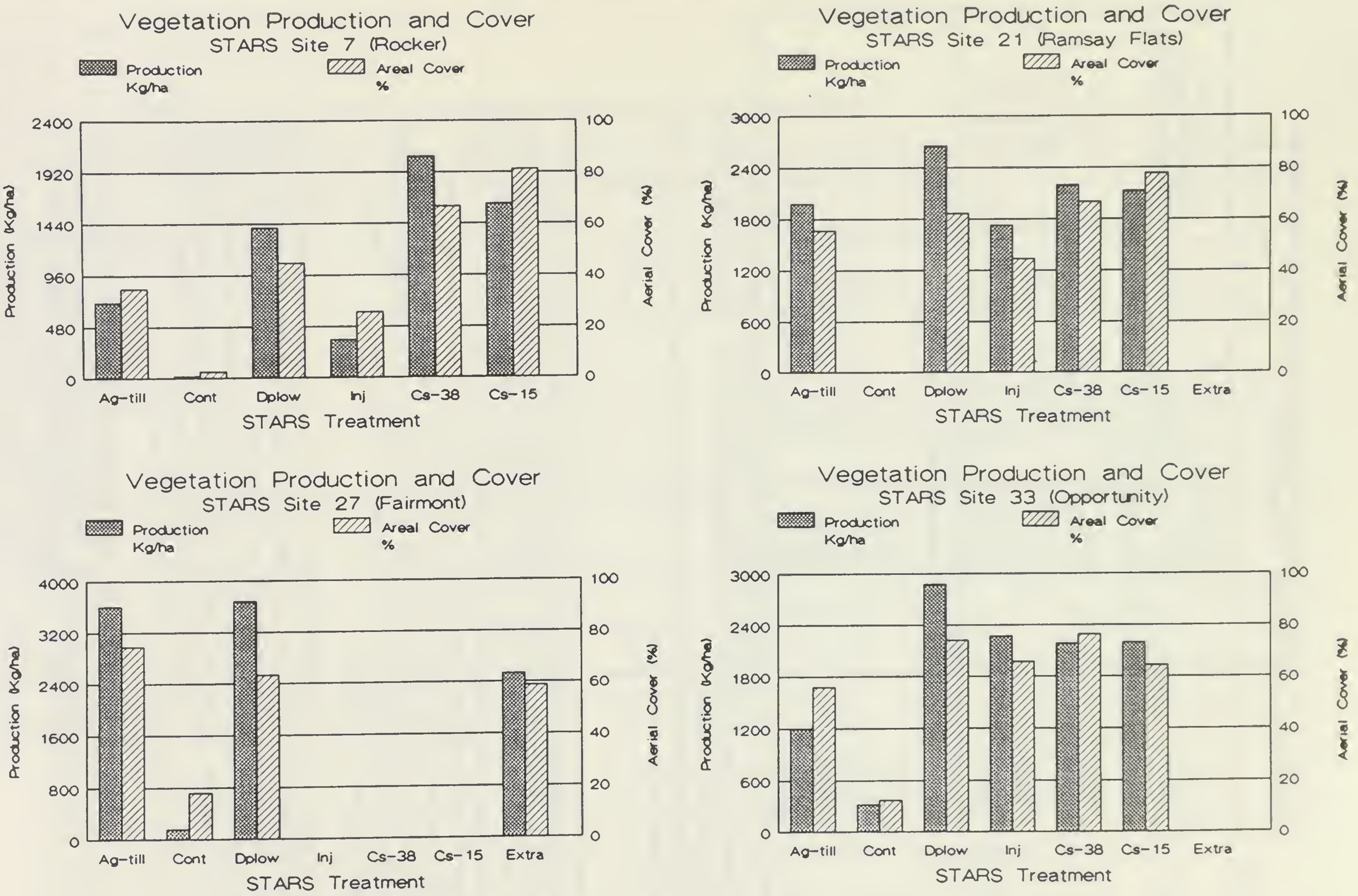

Figure 5. Vegetation cover and production at STARS sites (1991).

Table 3. Comparison of mean elemental levels (mg/kg) in dominant vegetation collected (July 1991) from STARS Site 21 (Ramsay Flats).

	Treatment					
Element	**Agricultural Tillage**	**Control**	**Deep Plow**	**Slurry Injection**	**Topsoil**	**SE Montana Western Wheatgrass[2] (*Agropyron Smithii*)**
			Tall wheatgrass (Agropyron elongatum)			
Aluminum	316.0 AB[1]	[2]	384.0 B	302.0 AB	74.0 A	5 to 50
Arsenic	26.1 AB		33.2 B	21.9 AB	4.3 A	<1 to 2
Cadmium	2.0 B		1.7 AB	1.5 AB	0.61 A	0.01 to 0.1
Copper	278.0 B		247.0 AB	216.0 AB	63.9 A	2.5 to 4.5
Lead	30.5 AB		60.6 B	25.1 AB	5.4 A	1.0 to 2.0
Manganese	917.0 B		672.0 AB	512.0 A	320.0 A	25 to 65
Zinc	750.0 B		541.0 AB	393.0 A	249.0 A	12 to 21
			Intermediate wheatgrass (*Agropyron intermedium*)			
Aluminum	311.0 B	[3]	266.0 B	286.0 B	48.4 A	
Arsenic	21.7 B		22.3 B	26.6 B	3.1 A	
Cadmium	1.9 B		1.6 AB	2.1 B	0.57 A	
Copper	228.0 B		196.0 AB	268.0 B	59.5 A	
Lead	28.3 B		38.2 B	29.7 B	3.5 A	
Manganese	790.0 B		622.0 B	677.0 B	243.0 A	
Zinc	589.0 B		491.0 AB	575.0 B	195.0 A	

[1] Multiple mean comparison based on LSD at significance level of 0.05. Means followed by same letter in *rows* are not different.

[2] Al and As estimated from the literature, Cd, Cu, Pb, Mn, and Zn from Munshower et al. 1987).

[3] Species did not grow on control plot.

These data were collected at regular intervals in 1990, 1991 and 1992, with the purpose of developing water budgets for each STARS treatment and for untreated streambank tailings. Development of a water budget quantified the efforts of treatment on infiltration runoff, leachate generation, and plant water use. Lack of space for this paper prevents a full discussion of the soil hydrology monitoring results, water budgets and modeling efforts for long-term impact contaminant movement. These are fully described in the STARS Phase III Final Summary Report (Schafer and Associates and Reclamation Research Unit 1993). Modeling of the fate and transport of metals in surface waters at Silver Bow Creek is presented in a companion document in this symposium (Schafer et al. 1993). For this paper, a brief presentation of simulated rainfall/runoff test results is presented.

The study area has a history of fish kills resulting from the erosion of metals, salts, and acid into the receiving Clark Fork River after precipitation events of sufficient magnitude to wash mining wastes into the river. It has been assumed that amendment of the materials and vegetation growing on them will increase infiltration, leach salts deeper in the profile and reduce erosion of surface salts. During the third growing season simulated rainfall was applied to several plots to evaluate this phenomenon. Precipitation rates were 5 cm/hr for two hours. Runoff at three sites was collected, measured, and elemental levels in the water determined.

Coarse textured material at Site 7 had relatively little runoff, especially after amendment application and plant establishment. Deep plow and agricultural tillage treatments yielded insufficient runoff for chemical analyses. Runoff at Site 21 (Table 4) was adequate from all

Table 4. Contaminants in runoff water from simulated rainfall[1] tests, Site 21 (Ramsay Flats).

Analyte Mass (mg)	Treatment and Total Runoff Volume				
	Ag-Till (2.84 l)	Control (20.15 l)	Deep Plow (7.00 l)	Injection (16.10 l)	Coversoil (0 l)[2]
Arsenic					
Dissolved	0.11	1.08	0.33	1.48	0
Copper					
Total	34.1	676	11.6	73.2	0
Dissolved	0.41	579	2.8	0.78	0
Iron					
Total	119	1200	38.2	222	0
Dissolved	0.17	4.32	0.13	0.50	0
Manganese					
Total	37.6	2120	56.7	83.5	0
Dissolved	14.2	2050	52.1	0.87	0
Zinc					
Total	35.5	1680	30.5	83.9	0
Dissolved	1.46	1590	20.47	0.41	0

[1] Simulated rainfall rate was 5 cm/hour for a period of 2 hours.

[2] No runoff was obtained from coversoil treatment plot.

treatments for the analytical determination of the elemental levels. Results of the analyses support the hypothesis that amendment application will reduce contaminant concentrations entering surface waters. Table 4 shows the remarkable decreases in metal mobility in runoff from all treatments at Site 21.

CONCLUSIONS

Lime amendment of acid generating mining wastes has been shown to reduce the phytotoxicity of the materials but incorporation of the amendment into the wastes has been and continues to be a major problem. This study evaluated amendment and incorporation of wastes into several unique types of mining wastes. Amendments consisted of calcium carbonate and calcium hydroxide but the amount of the amendment added to each study site varied with the properties of the individual sites. Incorporation techniques included slurry pressure injection, agricultural plowing, deep plowing, and covering the waste with a soil cap. Several parameters were measured to determine the effectiveness of each incorporation technique.

While capping any waste with soil is a preferred rehabilitation technique, expense and lack of suitable materials often prevents the use of this technique. Of the alternative incorporation techniques evaluated in this treatability study, the deep plow appeared to produce the most satisfactory response in terms of ameliorating soil chemistry and soil hydrology, and in providing an acceptable environment for plants to grow and reproduce.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium 1993

MODELING THE FATE AND TRANSPORT OF METALS IN SURFACE WATER AT THE SILVER BOW CREEK CERCLA SITE

by:
William M. Schafer[1], J.D. Goering[1], T.R. Grady[1], E. Spotts[1], and D.R. Neuman[2]

Abstract

In-situ remediation of mining waste deposited along the margins of Silver Bow Creek was evaluated in a multi-year research project funded cooperatively by the State of Montana, EPA, and ARCO Coal. The objective of the Streambank Tailings and Revegetation Study (STARS) was to evaluate the environmental performance of base amendment addition, deep mixing techniques, and revegetation on the fate and transport of key metals of concern at the site.

This paper summarizes a portion of the STARS project relating to the effect of lime amendments and revegetation on runoff and erosion from streambank areas contaminated with tailings. In addition, the consequence of metals migration into surface water during high-intensity summer thunderstorms was evaluated for both the existing as well as treated tailings. The USDA-ARS GLEAMS model was used to predict long-term runoff and erosion from the site.

A three-year GLEAMS simulation indicated that STARS treatments would decrease runoff by 2 to 3-fold, and would change the timing of runoff. On the existing tailings, runoff was predicted intermittently from March through September. On reclaimed areas, flows were only expected in March and April when Silver Bow Creek provides more dilution. Substantial reductions in metal loading due to runoff from mid-summer thunderstorms (historically associated with fish kills) were also predicted.

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[1] - Schafer and Associates, Bozeman, MT. 59715
[2] - Reclamation Research Unit, Montana State University, Bozeman, MT 59717

INTRODUCTION

The Silver Bow Creek site located downstream from the Butte mining complex is the largest CERCLA mining waste site in the nation. Large volumes of sulfide tailings have been fluvially transported in Silver Bow Creek and the upper Clark Fork river. As a consequence, the floodplain system is widely contaminated with metal-enriched low pH materials which have contributed to degradation of surface water quality.

The STARS investigation was initiated to develop *in-situ* methods for remediating streambank tailings areas. A series of small plots were amended with lime and seeded with metal-tolerant species in 1989. Various amendment incorporation methods were evaluated in the field investigation. Environmental performance monitoring of the STARS treatments was conducted from 1989 through mid-1992 (Neuman et al. 1993 this publication, Schafer & Associates and Reclamation Research, 1989a and 1989b). The purpose of this investigation is to report on the effectiveness of selected STARS treatments in reducing the transport of metals into Silver Bow Creek via surface runoff and erosion.

METHODS

GLEAMS Model Theory and Structure

The USDA-ARS GLEAMS model was used to simulate runoff and erosion from STARS treatments compared to untreated streambank tailings. The GLEAMS model consists of several component submodels for calculation of root zone hydrology, erosion, nutrient flux, and pesticide flux. Separate parameter input files are developed to run each component. Pass files are created by each component run for use in subsequent batch routines. Only the hydrology and erosion components of the GLEAMS model were used for this simulation.

The first component, hydrology, uses daily rainfall data, monthly temperature and solar radiation data, and various soil parameters for computation of the daily water balance. The amount and timing of runoff as well as other components of the water balance are computed, and pertinent information on storm size and runoff are passed to the erosion model component.

The technique used for estimation of runoff from daily rainfall data is the SCS curve number approach which has been widely adopted throughout North America. The curve number approach (Mockus 1985) relates the depth of runoff for a given depth of rainfall to the antecedent soil water content and to the "curve number". Curve numbers vary from 0 to 100, and are related to the infiltration capacity of the soil. Detailed guides have been developed for estimating curve numbers (Mockus, 1985)

Runoff is numerically related to runoff by [1]. During a rainfall event, no runoff is presumed to occur until the rainfall depth exceeds 0.2 of the remaining soil water storage capacity. Soil storage [2] is related to the relative degree of soil saturation (which is influenced by historical rainfall and evaporation) and by the maximum storage [3] which is inversely proportional to the curve number for condition I. Condition I is the curve number for dry soil moisture conditions. While curve numbers for condition II (average soil moisture) are tabulated in USDA references, CN(I) values corresponding to CN(II) values can be calculated using [4].

$$Q = \left[\frac{(P-0.2s)^2}{(P+0.8s)}\right] \qquad [1]$$

where Q = *Runoff depth (inches)*
P = *Rainfall depth (inches)*
s = *soil storage coefficient (inches)*

$$s = s_{\max}\left[\frac{(UL-SM)}{UL}\right] \qquad [2]$$

where $s_{\max}$ = *see equation* [3]
UL = *Upper limit of soil water storage (inches)*
SM = *Current stored soil moisture (inches)*

$$s_{\max} = \left[\frac{1000}{CN(I)}-10\right] \qquad [3]$$

$$CN(I) = -16.91 + 1.348[CN(II)] - 0.01379[CN(II)^2] + 0.0001177[CN(II)^3] \qquad [4]$$

Runoff volumes for increasing rainfall amounts are shown in Figure 1. High curve numbers are typical of impermeable soils and disturbed areas while low curve numbers are typical of well-vegetated permeable natural soils. Implicit in the SCS curve number method is that runoff initiation occurs after 0.2 times the storage. In many urban areas where pavement and bare soils are common, runoff occurs during much smaller rainfall events. In addition, infiltration is assumed to decrease to zero after a rainfall depth equivalent to s has occurred. This assumption too may be erroneous especially for permeable soils.

Nonetheless, the SCS curve number method has been widely validated using research watersheds from throughout the U.S.

While the SCS curve number approach relies on a generalized assessment of the antecedent moisture condition, the GLEAMS model maintains a daily water balance so that s can be computed. In addition, the water balance module can keep track of cumulative runoff, evaporation, transpiration, percolation, and changes in soil water content [5].

Daily evaporation is the sum of soil and plant evaporation. The maximum daily potential evaporation (PET) [6] is based on the mean daily temperature and solar radiation, each of which are input as mean monthly values in the GLEAMS model. The PET calculated by the Ritchie equation is generally higher than that calculated by the solar thermal unit equation selected for PET calculation in the STARS work plan (Schafer and Associates and MSU, 1990).

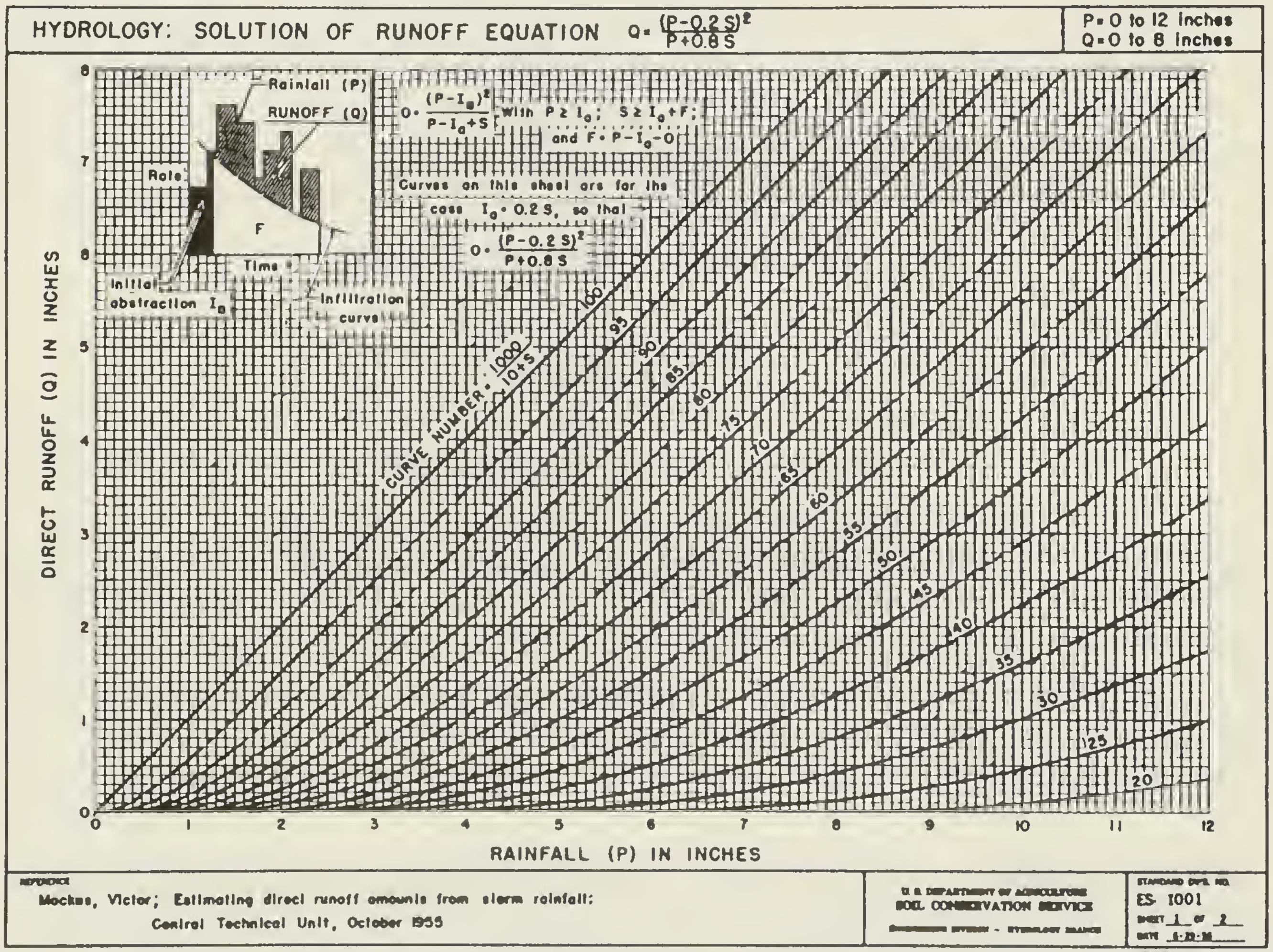

Figure 1. **Rainfall-runoff relationships for soils with varying curve numbers (from SCS National Engineering Handbook, Section 4).**

Soil evaporation [7] is calculated as a negative exponential function of the leaf area index (LAI). Hence, as the plant canopy coverage increases, soil evaporation decreases in importance. Prior research on soil evaporation from bare soil surfaces (Hillel, 1980) has shown that it occurs as a two-stage process. When the soil surface has a high water content, soil evaporation continues at a rate limited by the potential evaporation (E_o). When the upper few millimeters of the soil surface dries, however, the soil evaporation rate decreases due to the slow rate of vapor phase transmission of water. The two-stage soil evaporation process utilized by the GLEAMS model is given in [8]. Values for α_s for different soil textures are given in the GLEAMS manual. When the calculated soil evaporation exceeds ϵ_s, the slower stage of soil evaporation is triggered.

$$P = RO - ET - PERC \pm SOIL\ WATER \qquad [5]$$

$$E_o = \left[\frac{1.28\ \delta\ \eta_o}{\delta\ \gamma}\right]$$

$$E_o = \textit{potential daily evaporation (inches)}$$

$$\delta = \left[\frac{5304}{T^2}\right] e^{21.255 - \frac{5340}{T}}, T \textit{ in degrees Kelvin} \qquad [6]$$

$$\eta_o = \frac{(1-\lambda)(R)}{58.3}$$

$$\lambda = \textit{albedo}$$

$$R = \textit{solar radiation (langleys/day)}$$

$$\gamma = \textit{psychrometric constant}$$

$$E_s = E_o\ e^{-0.4\ LAI} \qquad [7]$$

Plant evapotranspiration is assumed to be a linear-weighted fraction of the PET for LAI values up to 3.0. When soil water is limited (less than 25 percent of available water), the plant evaporation decreases linearly until the soil is dry. Values of LAI can either be provided by the GLEAMS pre-processor for over 75 agronomic crop and forage species or can be input as a function of growing season length and julian date. LAI values of zero default to simple bare soil evaporation. Soil water utilized by plants is extracted preferentially from upper soil layers. As the growing season progresses, water extraction can occur at progressively greater depth. This algorithm simulates root extension of annual crops and root activity of perennial crops.

The erosion component of the GLEAMS model utilizes the Yalin equation to compute the sediment transport capacity of runoff. Estimated soil loss can either be "detachment-limited" or "transport-limited". Detachment can occur either due to rain-drop splash or due to the energy of overland flow.

$$\begin{gathered}
E_s = E_o e^{-0.4LAI} \ \mathit{where}\ \epsilon_s < E_o \\
E_s = \alpha_s\left[t^{0.5} - \left(t - \frac{1}{2}\right)^{0.5}\right] \ \mathit{where}\ \epsilon_s > E_o \\
\epsilon_s = 9(\alpha_s - 3)^{0.42} \\
\epsilon_s = \mathit{maximum\ daily\ soil\ evaporation} \\
\alpha_s = \mathit{evaporation\ coefficient}
\end{gathered} \tag*{[8]}$$

$$\begin{gathered}
E_p = \left[\frac{(E_o\ LAI)}{3}\right] \ \mathit{for}\ LAI<3 \\
E_p = E_o - E_s \ \mathit{for}\ LAI>3
\end{gathered} \tag*{[9]}$$

Overland flow from hillslopes can be routed in a number of ways. The simplest simulation is for overland flow to be routed directly to a channel at the edge of the domain being modeled. In addition, overland flow can be concentrated into a channel within the field boundary. Finally, the hillslope profile can be segmented into various shapes (*eg.* uniform, convex, concave, or complex). Overland flow and sediment is routed through each segment. Deposition or flow-induced detachment can occur within any hillslope of channel element within the model domain.

The GLEAMS model also predicts the particle size distribution of eroded sediment. When deposition occurs within the field area or channel or when transport capacity is exceeded the transported sediment may consist on average of finer particles than the average grain size in surface soils.

GLEAMS Model Validation

The hydrology and erosion components of the GLEAMS model have been validated at experimental research watersheds in Montana, Texas, Oklahoma, Ohio, Georgia, Nebraska, West Virginia, Mississippi, Iowa, Arizona, and New Mexico. In general, the long-term trend in runoff and erosion rates were accurately predicted by GLEAMS although model performance on individual storms was less reliable.

MODEL CALIBRATION

Parameter Estimation: The GLEAMS model requires calibration before it can be used to simulate runoff and erosion from unknown areas. Due to the complexity of the GLEAMS model, only the control and the most successful amendment/revegetation treatment at each location were simulated using GLEAMS.

Precipitation input for GLEAMS runs was from the Anaconda, MT NOAA weather station (average precipitation 12.8 inches). A comparison of rain gauge measurements from each of the STARS field sites indicated that the Anaconda site correlated well with other stations in the basin and tended to have somewhat higher cumulative precipitation. Three full years of input data (1989 through 1991) were used in the GLEAMS simulations. Mean monthly average temperature and solar radiation values were calculated for the Ramsay Flats climate station for the entire period of record. GLEAMS results are not highly sensitive to small variations in daily temperature and solar radiation, hence monthly inputs were used.

Input parameters for the hydrology and erosion simulation are listed in Table 1. The most sensitive parameter in the GLEAMS model is the runoff curve number. Due to the sensitivity of this parameter, great care was taken in estimating CN values. Rainfall simulation test data were used to calculate CN values. The control plots at Ramsay Flats and Opportunity had unusually high curve numbers presumably because of the rainfall-induced compaction on the exposed tailings. A curve number of 95 was used for the control plots at each site. The coarse texture of tailings at Rocker resulted in a control curve number estimated to be 55. Measured curve numbers for Ramsay Flats and Opportunity were 94 and 93 while the Rocker site had a CN of less than 50. Due to the extremely high infiltration rate, no runoff occurred at Rocker during the rainfall simulation tests. The deep-till/revegetated plots at Ramsay Flats and Opportunity had much lower measured CN values than the control (59 and less than 50) due to the effects of tillage and revegetation. The curve numbers selected for the GLEAMS simulation (78 at Ramsay Flats and 65 at Opportunity) were conservatively set higher than the measured CN values due to potential bias in site selection for the rainfall simulation tests.

In general, a relatively thin root zone depth was input to simulate the control plots because observed changes in soil water content due to evaporation were confined to the upper 12 inches of soil. Values for hydraulic conductivity, and soil water-holding capacity were based on measurements taken at each site.

The leaf area index (LAI) values and the crop type are important variables in simulating the on-site water balance. For the control plots, a LAI value of near zero was input so that GLEAMS would default to soil evaporation. For the vegetated plots, numerous preliminary GLEAMS runs were performed to try to simulate the observed changes in soil water content from the site. Very high values of LAI had to be input in order for plant evapotranspiration to approach soil evaporation in magnitude. Despite the fact that actual LAI values on the deep-till plots are near 1 or less, higher LAI values were input to improve model results. The beginning and end of the growing season and relative seasonal LAI values were input to simulate actual conditions.

Table 1. Input parameters used for the GLEAMS model - Silver Bow Creek, Montana.

GLEAMS MODEL DOMAIN	
HYDROLOGIC PARAMETERS	EROSION PARAMETERS
Area = 1,000 by 1,000 trapezoidal watershed Saturated Hydraulic Conductivity = measured at each site Soil Profile Percent Full = Initial water content 85% of field capacity Evaporation Coefficient = varies by site SCS Curve Number = varies by site, based on rainfall simulation Root Zone Depth = varies by site, generally 6 inches for control and 36 inches for vegetated Soil Characteristics = porosity, field capacity, wilting point, organic matter percent, clay, and silt content varies by site, based on observed soil morphology Monthly Mean Daily Maximum and Minimum Temperature = Based on Ramsay Flats climate station Monthly Mean Daily Solar Radiation = Based on Ramsay Flats climate station Vegetation Characteristics = varies by site, growing season duration and LAI based on calibration results	Slope = Overall 1.5 %, 1,440 feet at 1.5 % then steepening to 30% for last 30 feet Watershed shape = length to width ratio 2:1 Soil Erodibility K Factor = varies by site, based on USLE nomograph Cropping Practice P Factor = set to 1.0 for all simulations Cover Factor = varies by site, generally 1.0 for control and 0.2 for vegetated

SIMULATION MODEL RUNS

Ramsay Flats: Runoff and erosion was simulated for the control and deep plow plots at Ramsay Flats using the calibrated GLEAMS model (Table 2). Significant differences in the amount of runoff and erosion were predicted between the control and deep plow treatments. For the three year simulation, 1.46 inches of runoff occurred from the control while only 0.67 inches was predicted from the deep plow plot. No percolation below the root zone was predicted for either treatment, hence evaporation and transpiration accounted for the remaining average annual rainfall of 12.8 inches.

The timing of runoff also differed between the control and deep plow treatments. Peak monthly runoff for both treatments occurred in April or May depending on the year.

Measurable amounts of runoff occurred throughout the summer from the control plot while no runoff occurred after the end of May from the deep plow plot. This difference in the timing of runoff is thought to be significant in that most fish kills on the Clark Fork have been observed in July and August after convective thunderstorms. Runoff during midsummer may contain higher concentrations of dissolved metals due to the formation of metal-enriched salt crusts in the soil surface during warm weather. In addition, the higher spring instream flow means that runoff from streambank areas is more diluted when it mixes with the channel in spring than in summer.

Significant differences in erosion rates were also noted between the control and deep plow treatments. For the control site, an average annual soil loss of 13.7 tons/acre was predicted while only 1.2 tons/acre was predicted for the deep plow site. The ten-fold reduction in erosion was due to the reduction in runoff as well as the protection provided by the vegetative cover established on the deep plow plots.

Rocker: Similar simulations of the control and deep plow plots at Rocker were also conducted (Table 5.2). The soil material at Rocker was coarser in texture than at Ramsay Flats, hence had a lower water-holding capacity, a much higher infiltration rate, and had less erosive soils than at Ramsay Flats. The water balance results for three years of simulated rainfall at Rocker indicated that 1.69 inches of percolation (*eg.* groundwater recharge) would occur on the control plot, while the higher evaporative use of water by established vegetation in the deep plow plot would prevent percolation. Runoff averaged 0.66 inches and 0.61 inches on the control and deep plow plots respectively. Predicted runoff was less at Rocker than at Ramsay Flats due to the lower runoff curve number of both the control and deep-plow plots at Rocker. Runoff was only predicted in March and April, presumably in response to snowmelt events. Less difference in runoff was noted between the control and deep plow plots because reclamation had less overall effect on the curve number (CN = 40 for both cases) due to the already rapid infiltration on the control plot.

Predicted soil loss at the Rocker site was 0.7 tons/acre for the control and 0.14 tons/acre for the deep plow plot. These low rates of soil loss were due to the coarse texture and rapid infiltration rates of soil at Rocker.

Opportunity: GLEAMS simulation of the Opportunity control site (Table 5.2) yielded an estimated annual 1.69 inches of runoff and 0.42 inches of percolation for 12.8 inches of precipitation. Runoff occurred intermittently from March through September with peak contribution in April. Percolation occurred from April through June when the soil profile was at its annual maximum.

The deep plow plot at Opportunity had substantially less predicted runoff (0.66 inches/year) than the control plot and no percolation. Runoff occurred only during snowmelt in March and April, periods of the year when runoff would be expected to have less impact on fisheries and macroinvertebrates due to the higher in-stream flows. The decrease in runoff was due to the reduction in the curve number observed between bare tailings (curve number = 94) and the deep-plowed and vegetated condition (curve number = 65).

Predicted annual soil loss for the control plot and deep plow treatment was 9.2 and 0.67 tons/acre respectively.

Table 2. **Summary of water balance results and soil loss from the USDA CREAMS runoff model for the Rocker and Opportunity sites Flats (1989 to 1991).**

WATER BALANCE TERM	RAMSAY CONTROL	RAMSAY DEEP PLOW	ROCKER CONTROL	ROCKER DEEP-TILL	OPPORTUNITY CONTROL	OPPORTUNITY DEEP-TILL
WATER BALANCE SUMMARY						
Precipitation (in/yr)	12.8	12.8	12.8	12.8	12.8	12.8
Runoff (in/yr)	1.46	0.67	0.66	0.61	1.69	0.32
ET (in/yr)	11.8	14.1	10.6	12.9	10.9	12.3
Percolation (in/yr)	0.0	0.0	1.69	0.0	0.42	0.0
SOIL LOSS SUMMARY						
Soil Loss (t/acre/yr)	13.7	1.2	0.7	0.14	9.2	0.67

FATE AND TRANSPORT ASSESSMENT

Hydrograph Mass Loading: Runoff rates predicted by the GLEAMS simulation were compared with USGS streamflow records at Opportunity to determine the mass loading of runoff from streambank areas compared with instream flows. In addition, chemical characteristics of water collected during rainfall-runoff trials were used to calculate mass loading of metals into Silver Bow Creek during typical high-intensity thunderstorms. The purpose of this fate and transport analysis was to;

- determine if modeling results identify surface runoff of metals as a critical transport mechanism;
- evaluate the effectiveness of STARS treatments as a means of reducing potential impacts to surface water;
- and to determine the relative importance of dissolved versus total metals delivered to Silver Bow Creek.

In order to compute the mass loading of metals in surface runoff from streambank areas, it was first necessary to estimate the contribution of runoff water from streambank areas. For this analysis, the contributing area was assumed to be the lower 10 miles of the Silver Bow Creek floodplain. Only the lower portion of the floodplain was used because most convective storms are rather localized, and it was implausible to assume that the entire watershed would be affected by a convective storm cell. The streambank tailings contributing area was assumed to consist of a 100 foot wide corridor (120.8 acres) along Silver Bow Creek. Rainfall-runoff characteristics from the Opportunity control plot were used to simulate the response of the streambank tailings area. Runoff from the streambank tailings area was assumed to have zero time of concentration due to its close proximity to the channel. Duration of each daily runoff event was assumed to be 6 hours.

Actual flows from the USGS gauging station (#1232600) at Opportunity were used to determine the amount of dilution which would occur when streambank tailings runoff reached Silver Bow Creek. The entire three period of record used in the GLEAMS simulation run was reviewed, and three time periods were selected representing the single largest runoff event from each year. In 1989 (Figure 2), a 0.80 inch rainstorm on 8/23 resulted in 0.11 inches of runoff. This event over a 6-hour duration resulted in 2.23 cfs of flow from the streambank area which would have made up 8.9% of the instream flow of 25 cfs reported for 8/23. As expected, the tailings area runoff peak from the 8/23 event occurred on the rising limb of the basin-wide hydrograph. This response would be expected due to size of the Silver Bow Creek watershed at Opportunity (284 mi^2).

Similar analysis of stream hydrograph records and computed runoff in May, 1991 (Figure 3) and in September, 1991 (Figure 4) resulted in streambank runoff computed to be 15.1 % and 9.2 % of baseflow.

Metal Mass Loading: Runoff from simulated rainfall tests from untreated and lime-amended plots at Opportunity were analyzed for a suite of dissolved and total metals. Table 3 lists the levels of dissolved and total copper and zinc in runoff samples representing the "first flush" of runoff (0.05 inch) as well as the long-term runoff. Assuming that runoff from streambank tailings can comprise from 10 to 15 % of the total flow in Silver Bow Creek during an intense thunderstorm, the calculated level of total copper and zinc may range from 1.4 to 9.0 mg/l and 1.1 to 6.9 mg/l respectively. Dissolved copper and zinc may range from 0.4 to 4.5 mg/l and 0.93 to 5.25 mg/l respectively. In addition, the predicted tailings erosion

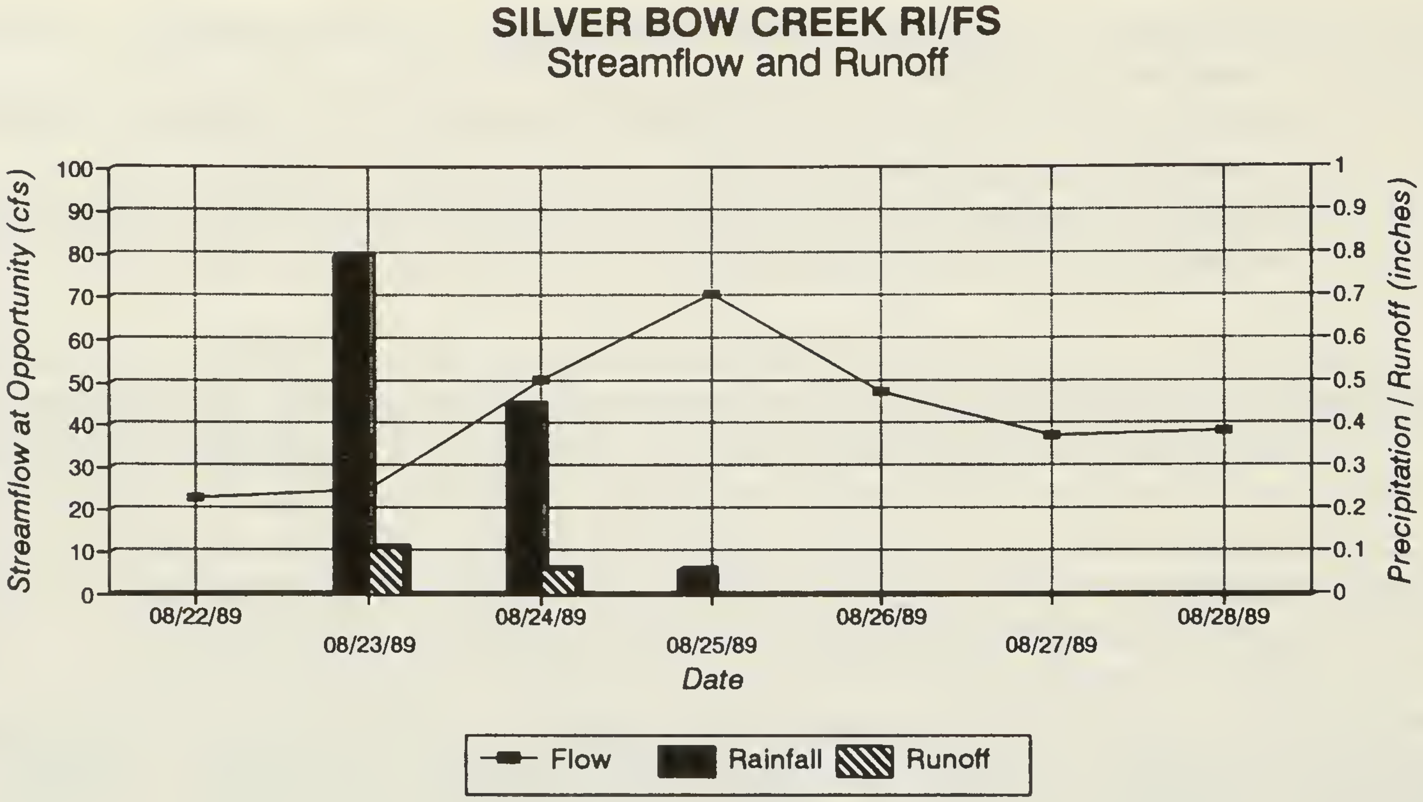

Figure 2. Comparison of daily rainfall and estimated streambank tailings runoff with Silver Bow Creek flow in August, 1989.

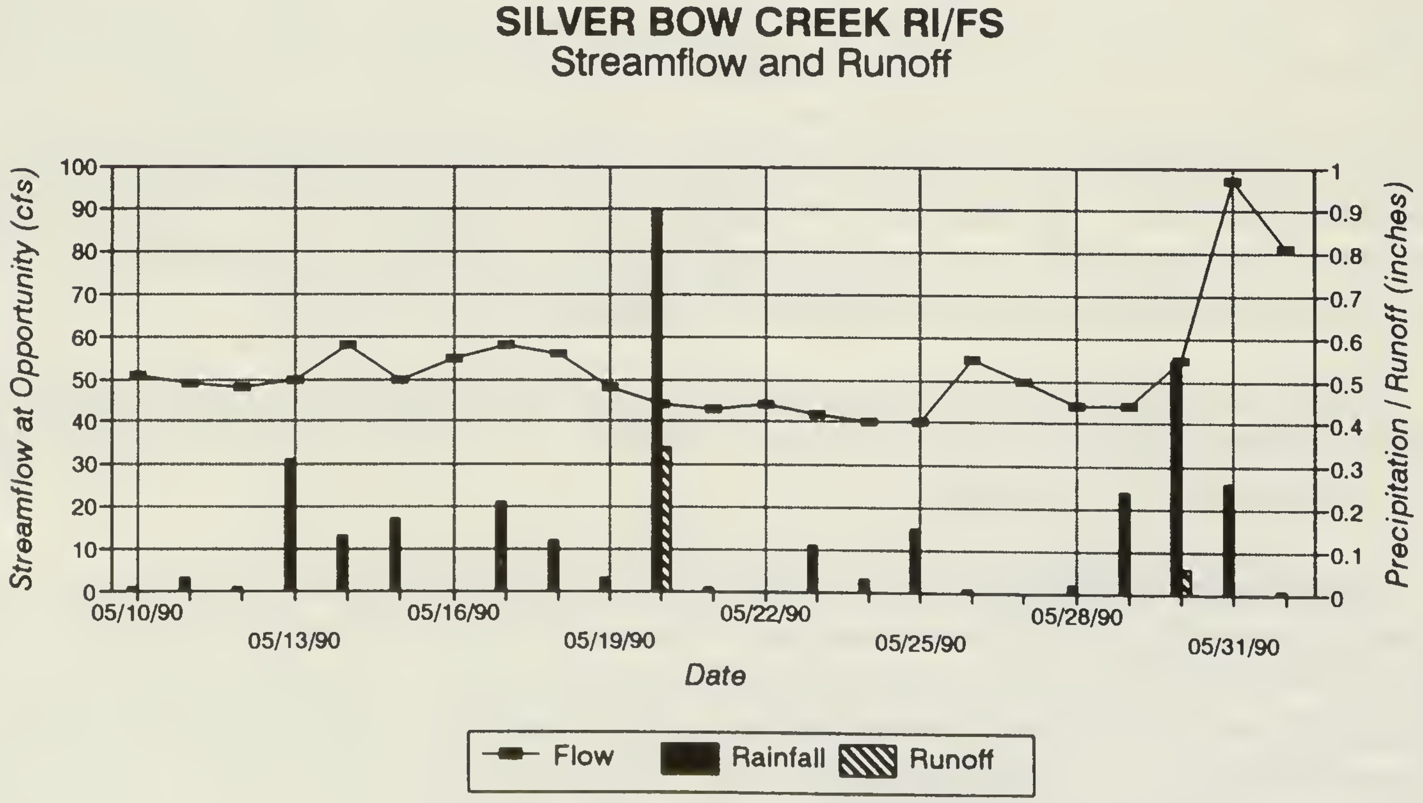

Figure 3. Comparison of daily rainfall and estimated streambank tailings runoff with Silver Bow Creek flow in May, 1990.

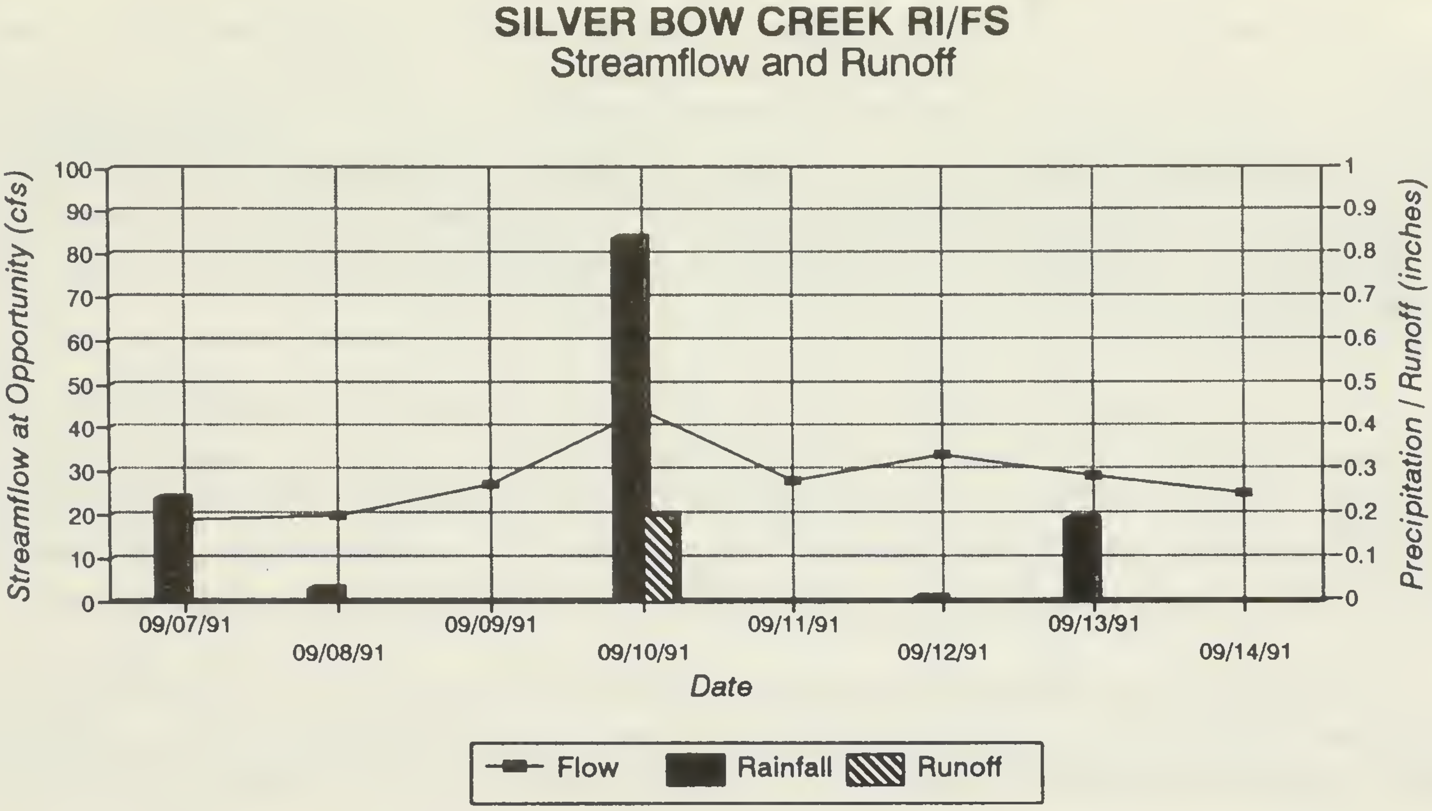

Figure 4. **Comparison of daily rainfall and estimated streambank tailings runoff with Silver Bow Creek flow in September, 1991.**

rate for these runoff events would be expected to deliver an equal amount of metals to Silver Bow Creek to those predicted on the basis of runoff test results. Hence, total metal loads could be twice the concentrations stated. These levels of copper and zinc exceed the acute criteria for protection of freshwater life by many orders of magnitude. The hardness-corrected copper and zinc standards at a hardness of 100 mg/l are 0.018 and 0.117 mg/l. Clearly, runoff from streambank tailings areas is a critical metal migration route in the existing Silver Bow Creek system.

The runoff contribution of metals to Silver Bow Creek from revegetated plots was also calculated. On each of the dates when runoff was simulated from bare tailings, no runoff was predicted from any of the lime-amended and vegetated plots. As a consequence, predicted metal loading was decreased to zero by the STARS treatments. It should be noted that while no runoff was predicted for convective thunderstorms during the summer months, runoff was expected to occur during snowmelt. The maximum contribution of streambank runoff to Silver Bow Creek for revegetated plots occurred in March or April of the year and was equal to less than 2 percent of the instream flow. Coupled with the 25-fold or greater reduction in total metal concentration in runoff from revegetated plots, the overall contribution to Silver Bow Creek from the streambank areas would be expected to decline by 100 to 200-fold due to basin-wide implementation of the STARS technology.

The chemical speciation of metals in surface runoff (Table 4) show many interesting differences between the control and lime-amended plots. Runoff from the control plots consist of three chemical fractions: dissolved ions, relatively inert ions associated with the solid fraction (TSS fraction); and other ionic forms dominated by metal hydroxide precipitates and adsorbed phases. The biological "availability" is expected to differ between the three phases with dissolved being the most readily available and the TSS fraction being mostly unavailable.

Only 20 percent of the copper and zinc in runoff from the control plots was associated with the TSS fraction, with all of the remaining zinc in a dissolved form. The remaining copper consisted of nearly equal fractions of dissolved and precipitated forms. This difference in partitioning between copper and zinc would be expected due to the tendency for copper to precipitate from solution at lower equilibrium pH than zinc.

In runoff from the lime-amended treatments, 30 to 60 percent of the total metal load was in the TSS fraction. The remaining metals were equally divided between the dissolved and precipitated fraction. This change in metal partitioning from relatively available forms in runoff from control plots to more unavailable forms on treated plots further magnifies the 20 to 30-fold reduction in total metal levels in runoff from limed plots.

Table 3. Total and dissolved copper and zinc in simulated rainfall-induced surface from control and lime-treated plots at Opportunity. "Initial runoff" represents the first 0.05 inches of runoff while "long-term runoff" represents an average of the next 0.5 to 1.0 inch of runoff.

CONSTITUENT	CONTROL PLOT		LIME-TREATED PLOT	
	INITIAL RUNOFF (mg/l)	LONG-TERM RUNOFF (mg/l)	INITIAL RUNOFF (mg/l)	LONG-TERM RUNOFF (mg/l)
Copper - Total	60	14.3	1.31	0.60
Copper - dissolved	30	4.0	0.45	0.093
Zinc - Total	46.2	11.4	1.66	0.41
Zinc - dissolved	35	9.3	0.55	0.033

Table 4. **Speciation of metals in surface runoff from streambank tailings areas at Opportunity.**

CONSTITUENT	CONTROL PLOT		LIME-TREATED PLOT	
	INITIAL RUNOFF (% of Total)	LONG-TERM RUNOFF (% of Total)	INITIAL RUNOFF (% of Total)	LONG-TERM RUNOFF (% of Total)
Copper - Dissolved	50	30	35	15
Copper - TSS Fraction	20	20	40	30
Copper - Other	30	50	25	55
Zinc - Dissolved	80	80	35	10
Zinc - TSS Fraction	25	80	40	65
Zinc - Other	0	0	20	25

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ANACONDA REVEGETATION TREATABILITY STUDY

R.J Montgomery[1]

ABSTRACT

The Anaconda Revegetation Treatability Study (ARTS) is being conducted to demonstrate in-place treatments of mill tailings, smelter wastes and soils impacted by smelter emissions at the Anaconda Smelter National Priorities List Site in Anaconda, Montana. The study area is approximately 11,000 acres in size and has been divided into five subareas. These areas are: the Opportunity Tailings Ponds, the Anaconda Tailings Ponds, Smelter Hill, Old Works and Adjacent Sites.

For nearly 100 years, the sulfide ores mined in Butte, Montana were processed at the Anaconda Reduction Works in Anaconda, Montana. Mill tailings were contained in the subareas of the Anaconda and Opportunity Tailings Ponds. These are acidic tailings and typically contain arsenic, copper, cadmium, zinc and lead. The remaining subareas have been effected by smelting processes and smelter stack emissions. The soils at these locations contain elevated levels of arsenic, copper, cadmium, zinc and lead but are neutral in pH.

This study is being conducted in four phases. The first Phase includes reviews of reclamation literature, personal interviews and data searches to determine why reclamation practices in the past have been successful or unsuccessful. Revegetation attempts at other locations with problems similar to the Anaconda site will be evaluated. Areas of natural revegetation will be documented. This will include species present, extent of colonization and conditions present. Data will be collected to identify and characterize candidate demonstration sites within the five subareas. The second Phase includes laboratory and greenhouse tests which will be used to develop effective amendment/vegetation treatments that will be further tested in field scale demonstrations. Phase Three is the implementation of the field scale demonstrations. Phase Four will monitor the effectiveness of the treatments in reducing contaminant movement in the surface water runoff, air and bioaccumulation pathways.

ARTS is funded by ARCO and is being conducted by the Reclamation Research Unit at Montana State University under the direction of a Technical Committee comprised of representatives from ARCO, the Environmental Protection Agency and the Montana Department of Health and Environmental Sciences.

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[1] ARCO, 307 East Park, Anaconda, Montana.

INTRODUCTION

Ore processing facilities were constructed at Anaconda in 1884. The ores mined in Butte, Montana were shipped to Anaconda and processed using a variety of techniques during the nearly 100 years of operation. Inorganic wastes such as mill tailings, slag and flue dust are found at various locations on the site. These wastes contain arsenic and metals such as copper, cadmium, zinc and lead. In addition, soils throughout the site contain arsenic and metals due to aerial emissions from the milling and smelting process.

In 1983, the Anaconda Smelter Site was listed on the National Priorities List under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA). The site includes Smelter Hill, Old Works/East Anaconda Development, Anaconda Soils, Regional Water and Waste, and Flue Dust operable units. In 1988, the Environmental Protection Agency (EPA) in consultation with the Montana Department of Health and Environmental Sciences (MDHES) and the Atlantic Richfield Company (ARCO), reached agreement for performance of the Anaconda Smelter Remedial Investigation/Feasibility Study (RI/FS). The treatability study is being conducted pursuant to this agreement by Montana State University under the direction of a Technical Committee comprised of representatives of ARCO, EPA and MDHES.

TREATABILITY STUDY OBJECTIVES

CERCLA requires that remedial alternatives must be evaluated in relationship to a reduction of risks to human health and the environment. The evaluation is influenced by contaminants present, their concentration, volume of contaminated media and potential exposure pathways.

As part of this process, the primary objective of ARTS is to develop and evaluate land reclamation techniques for establishing vegetation on five selected subareas within the Anaconda Smelter NPL Site. This evaluation will consider the ability of the technologies to meet the following criteria for selection of a final remedy under CERCLA:

- Provide overall protection of human health and the environment;
- Comply with Applicable or Relevant and Appropriate Requirements (ARARS);
- Reduce through treatment, the mobility and toxicity or volume of waste materials;
- Be cost effective;
- Be implementable on a large scale;

- Provide both short and long-term effectiveness; minimize long-term maintenance; and

- Be acceptable to the State of Montana and the community.

In addition to these criteria this study will also evaluate these technologies ability to:

- Preserve or enhance area aesthetics.

The ARTS treatability study is being conducted in four phases. Phase I includes reclamation literature reviews and data assessments. Phase II includes laboratory and greenhouse investigations. Phase III is the implementation of the field scale demonstrations. Phase IV will monitor the effectiveness of the treatments in reducing contaminated movement in the surface water runoff, air and bioaccumulation pathways.

Each phase has specific objectives. Phase I has the following objectives:

- Review and summarize site-specific reclamation procedures used by ARCO and the Anaconda Minerals Company and postulate why the procedures were successful or unsuccessful;

- Evaluate revegetation attempts at other locations with characteristics similar to the Anaconda site;

- Document areas of natural revegetation and postulate responsible mechanisms; and

- Collect data to identify and characterize candidate demonstration sites within the five subareas.

Phase II has the following objectives:

- Develop and test amendments for each subarea that will enable contaminated soils and tailings to support vegetation;

- Evaluate native and commercial plant species to determine those best adapted to amended or non-amended soils and tailings materials; and

- Recommend amendment/vegetation treatments to be evaluated in the field demonstrations for each subarea.

Phase III has the following objectives:

- Test technologies identified in Phase I and Phase II under field conditions at specific sites in each subarea;

- Determine effective techniques for applying and incorporation of amendments on a large scale;

- Develop planting techniques for large scale revegetation;
- Determine the need for post-planting maintenance; and
- Evaluate the factors responsible for success or failure.

Phase IV has the following objectives:

- Evaluate the long-term effectiveness of treatments established in the field demonstrations;
- Evaluate contaminant loading on and in vegetation;
- Evaluate effectiveness of amendment incorporation; and
- Evaluate potential for wind and water erosion.

DESCRIPTION OF ARTS SUBAREAS

Five subareas in the Anaconda Smelter Site were identified by the ARTS technical committee (Figure 1). Each site is unique, differing in physical setting, geochemistry and physical characteristics. A brief description of each subarea is presented below.

Opportunity Ponds

The Opportunity Ponds cover an area of approximately 3,400 acres and contain approximately 234,000,000 tons of mill tailings from ore processing operations. The ponds were in operation from 1914 through 1980. The ores processed at the Anaconda Smelter Site were typically sulfide, and the mill tailings are acid producing when oxidized. The physical characteristics of the tailings vary considerably due to differences in the ores processed and the processing methods used. Grain sizes vary from coarse sand to fine clay. Arsenic and metals such as copper, lead, zinc, and cadmium are found in varying concentrations throughout the ponds.

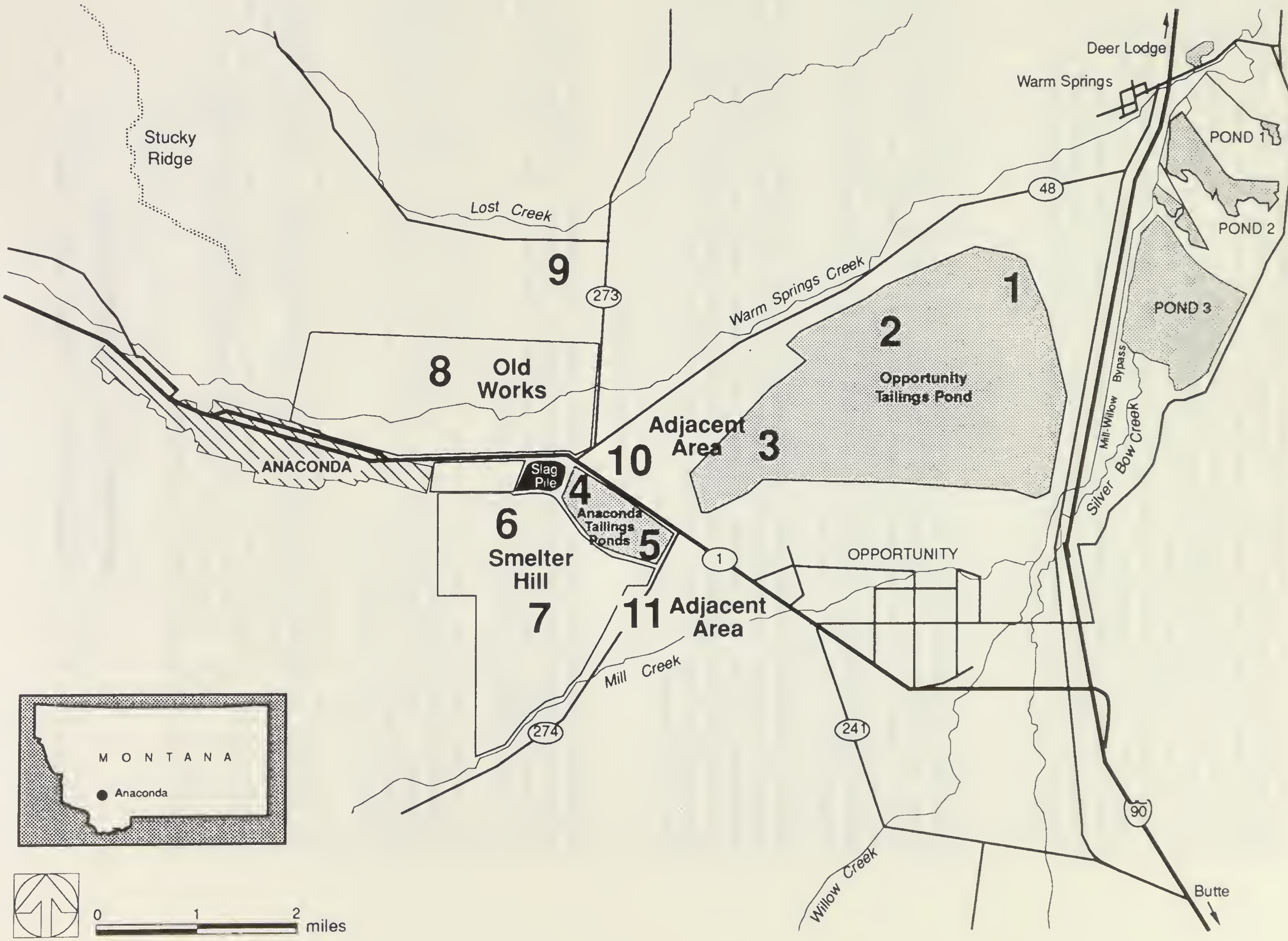
Anaconda Ponds

The Anaconda Ponds include an area of approximately 560 acres and contain approximately 103,000,000 tons of mill tailings. The oxidized zone of the tailings range from 3 to 6 feet and are acidic (pH 2 to 4). The grain sizes vary considerably from coarse sands to clays and contain metals similar to those found in the Opportunity Ponds.

SMELTER HILL

The ARTS study area on Smelter Hill is a 1,020 acre tract. This area includes the locations of concentrating, smelting and refining

PRELIMINARY ARTs DEMONSTRATION SITES



operations. The Anaconda Reduction Works began operations in 1902 and continued until 1980. Between 1982 and 1985 most structures were demolished and approximately 345 acres on the lower portions of the hill were revegetated by placing 12 to 18 inches of cover soil over a 100 to 400 ton per acre crushed lime stone layer. Many of these areas have been successful, while others are only sparsely vegetated. The vegetation has been monitored and the results are reported by Keammerer et. al. 1992. Natural revegetation is occurring in areas north, south and east of the stack. The area west of the stack is very steep (>50%) and supports little naturally established vegetation. The slopes north and east of the stack range from 25% to 33%.

Old Works

The Old Works is a 1,340 acre tract of milling and smelting wastes from the Upper and Lower Works which operated from 1884 to about 1902. Operations ceased when the Anaconda Reduction Works began operation. Present surface features at this site include demolition debris, mixed slag and jig tailings (Red Sands), heap roast slag and fluvially deposited tailings.

Natural revegetation has occurred in many areas of the Old Works. Trees, shrubs and grasses are established in the riparian areas adjacent to Warm Springs Creek and some species are beginning to establish themselves on the hillsides above the flood plain. Vegetation has also recovered on portions of the hillsides above the Old Works.

Adjacent Areas

Adjacent areas include lands near the Old Works, Smelter Hill and Anaconda/Opportunity Ponds where plant growth has been impacted primarily by wind and water deposited wastes. The adjacent areas include approximately 4,800 acres.

The area between the Old Works and Opportunity Ponds consists of contaminated soils that range from little or no visible tailings present, to tailings 1 to 18 inches in depth. Plant growth is often absent or consists of trees and shrubs with little or no grass or forb production.

South of the Opportunity Ponds, tailings are present on the surface at depths ranging from one inch to several feet. Plant growth is mainly deep rooted plants or shrubs. Southeast of the Anaconda Ponds plant growth may have been impacted by smelter stack emissions. In this area stands of young vigorous plants are found mainly in more protected areas. These plants appear to be expanding into the barren areas.

DEMONSTRATION SITE SELECTION

Preliminary field demonstration sites have been selected at locations that represent physical, geochemical, and agronomic limitations to revegetation within the five subareas. Initially 17 candidate sites were identified. Upon completion of field reconnaissance and site screening, 11 preliminary sites have been selected to conduct field demonstrations (Figure 1).

The objectives for screening of the candidate sites were:

- Determine chemical and physical characteristics for each candidate site so unique revegetation demonstration site can be identified; and

- Evaluate demonstration site variability.

Candidate sites were screened in two steps (Reclamation Research Unit, 1992). The first step was field measurements of pH, electric conductivity (EC), and texture. Soil profile descriptions were also made. Ten samples were collected from an evenly spaced grid within a three acre area of the candidate site. The variability in pH and salt content (as measured by EC) within the candidate site were assessed by calculating the mean value and 95% confidence levels associated with each mean. The confidence limit information for pH and EC and the soil textural information guided the first screening step for bulk sample collection. Once the confidence limits were established, three pits were excavated within the three acre area. Field screening tests for pH, EC and texture were conducted. The site screening tests had to be within the confidence limits defined by the grid sampling for the site to be of acceptable homogeneity for field demonstrations.

If a candidate site passed the first screening step, bulk samples were collected. In the second screening test representative subsamples of the bulk subsamples were analyzed for total sulfur, water soluble sulfur, HCL soluble sulfur, HNO_3 sulfur and the residual sulfur determined by the difference. If the sum of the HNO_3 sulfur and residual sulfur vary by less than 1.0% sulfur between the three pits, the site passed the second screening test. Variability of 1-4 % could lead to significant error regarding amendment rate and therefore lead to difficulty in interpreting treatment response variables.

If a candidate site passed screening tests 1 and 2, then bulk samples (three from each site) were analyzed for the following:

- pH, EC, organic matter, cation exchange capacity, and particle size distribution;

- Nutrient status, including nitrate-nitrogen, extractable phosphorus and potassium;

- Water soluble metals (Al, Cu, Fe, Mn and Zn), major cations (Ca, Mg, and Na) and water soluble As and B;

- Total metals (As, Cu, Fe, Mn, and Zn);
- Modified acid base account, including sulfur species, neutralization potential; and
- Exchangeable acidity.

Bulk samples from 11 sites were analyzed. The analytical results will be used to finalize the locations of demonstration sites within the five subareas.

PRELIMINARY ARTS DEMONSTRATION SITES

A description of the preliminary demonstration sites is given below.

Opportunity Ponds

Three locations were selected in the Opportunity Ponds. ARTS site No. 1 is located in the D-2 pond. This pond contains the largest quantity of unweathered pyrite in comparison to the other ponds in the complex since this was one of the last ponds used.

ARTS site No. 2 is located in the B-2 pond. This site is representative of conditions found in the B-1, C-1 and C-2 ponds. These tailings are generally acidic (pH <3) and coarse in texture.

ARTS site No. 3 is located in the A pond. This pond is the oldest pond in the area and has been dewatered for the longest period of time. These tailings have weathered for approximately 40 years resulting a very acidic, high sulfate material. These materials have oxidized and have exhausted nearly all of the acid production capacity near the surface (approximately 0-2 ft.).

Anaconda Ponds

Two sites were selected in the Anaconda Ponds. Site No. 4 is located on the perimeter of Pond 1 and Site No. 5 is located near the center of Pond 2. The difference between these sites is the particle size distribution with coarser materials being located on the pond perimeter and fine textured materials located toward the pond interior. These sites are very acidic (pH 2-4) and have high pyritic sulfur levels (2-5%). These sites will require specialized amendment applications and equipment to support plant growth.

Smelter Hill

Two sites were selected on Smelter Hill. Site 6 is located on the north side of Smelter Hill east of the buried flue. The slope is approximately 33% and the soils contain elevated levels of As, Cu and Zn. Site 7 is located on a 22% east slope of Smelter Hill. The steepness of the slope and some the highest concentrations of As, Cu and Zn found on the hill make this location unique.

Old Works

Three sites were selected in the Old Works area. Site 8 is located west of the Arbiter Plant where a veneer of rocky alluvium was place over the red sands slag and jig tailings. No plant growth is present. Site 9 is located east of the dragstrip where native soils have been eroded and covered, or simply covered with waste material which was either deposited aerially or fluvially. Deeper rooted shrubs and trees are found; however, grasses and forbs are limited to areas of fluvially deposited fresh soil along Warm Springs Creek.

Adjacent Areas

Site 10 is located west of the Opportunity A pond where fluvially deposited tailings have covered the native soil. The thickness of the tailings vary, from inches to several feet. Site 11 is located approximately 600 feet south east of the Anaconda Ponds. Windblown deposits from the Anaconda Ponds have contaminated the native alluvial soils. Plant growth in this area consists of islands of shrubs mostly devoid of grasses and forbs.

Final demonstration sites will be recommended by MSU in the Phase I Report. The Technical Committee will consider the recommendation and then select the final demonstration sites.

SUMMARY

At the time of this writing, Phase I is approximately 80% complete and the greenhouse portion of Phase II has begun. Field demonstrations are scheduled at the Smelter Hill and Old Works sites in 1993.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

TWO METHODS OF RECLAIMING STREAM ECOSYSTEMS IMPACTED BY MINE TAILINGS

C. T. Stilwell[1]

ABSTRACT

Two projects are presented, which describe the design and techniques used in remediating mine tailings having been fluvially deposited within and along a stream's floodplain. The purpose of this presentation is to describe and compare the two separate and distinct remedial designs and associated implementation techniques employed in the two projects. The potential advantages, disadvantages, and applicability of two designs are also discussed.

These two case studies are being implemented as treatability studies associated with the Remedial Investigation/Feasibility Study (RI/FS) for the Superfund project of Silver Bow Creek near Butte, Montana. Project I is to be constructed this year along a one-half mile stretch of Silver Bow Creek near Rocker, Montana, east of Butte. Project II is to be constructed this year on a one-half mile stretch of the same creek near Opportunity, Montana, fifteen miles downstream of Project I. As treatability studies, the purpose of the two projects are to collect data on the cost, implementability, and effectiveness of their designs to reclaim an impacted stream floodplain. This information collected will be used to in the decision of how to remediate the entire length of Silver Bow Creek. These two methods will also be compared with the design and techniques used with the Clark Fork Demonstration Project, which has been described in another presentation.

Similarities of the two projects include their ultimate goal, which is to reclaim a stream reach impacted by mine tailings to sustainable natural riparian setting. The designs of the reclamation for both projects include lime-amending and/or removing metal-laden tailings, then revegetating the ground to neutralize the tailings' impact on the soil, vegetation, and water resources along the stream. The lime amendment technology used is consistent with the technology studied and developed under the Streambank Tailings and Revegetation Studies (STARS) performed by Montana State University's Reclamation Unit. Lastly, both projects will employ conventional earth-moving and other construction equipment to implement the design.

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[1] Atlantic Richfield Company, 307 E. Park, Anaconda, Montana.

The general sequence of construction for both projects are:

1. Clearing and grubbing of debris
2. Tailings excavation/relocation/removal
3. Backfilling with clean fill (Project I only)
4. Treatment of soils by lime application/incorporation
5. Final site grading
6. Streambank stabilization for erosion protection
7. Revegetation with mix of riparian species

Project I was proposed to test a design which combines STARS-type lime amendment and revegetation technology with partial removal and relocation of tailings. The criteria for removing tailings was to remove all tailings and replace with clean soil where groundwater was within thirty (30) inches or less from the surface. Effectively, the only areas fitting this criteria were directly adjacent to the creek. The reason for removing tailings under these conditions instead of lime-amending all tailings-impacted areas was evidence that STARS-type lime amendment/revegetation techniques are less effective where groundwater is near the root zone.

The construction of Project I requires the clearing and grubbing of the project area, excavating fluvially deposited sediments and mine tailings within a corridor along Silver Bow Creek, relocating those materials along the margins of the 100-year floodplain of the creek, and bringing in clean borrow material to reconstruct the excavated areas. The relocated materials will be placed in four areas. These materials will be placed in one foot thick lifts. Lime kiln dust will be added to each lift and incorporated into the soils with an agricultural tiller to reduce the mobility of metals present in the material. Soils located outside of the excavated corridor but within the 100-year floodplain will be amended with lime to depths of up to 48 inches by deep lime incorporation methods.

When the excavation, backfilling, and lime incorporation tasks are completed, the site will be subjected to a final grading to drain towards Silver Bow Creek. The railroad grade adjacent to the south boundary of the project will also be graded to direct runoff around the relocated tailings. Two sites at the margins of tailings relocations areas will be riprapped to demonstrate erosion protection methods for this study. Revegetation will consist of application of fertilizer and a seed mix containing several plant species found in natural riparian areas of Montana.

An advantage of the Project I design is use of on-site areas to relocate tailings excavated from adjacent to the creek. This prevents the need for off-site repositories and reduces project costs associated with longer and more complicated tailings transport. Another advantage is the minimal excavation limited to tailings immediately adjacent to the creek requires less clean fill material which is difficult to find in Montana.

Project II was intended to test a remedial design which primarily relies on removal of the metal-containing and acid generating tailings, lime-amending the remaining native soil surface, and revegetating. No metal concentration removal criteria is used, therefore, some elevated concentrations of metals are expected to remain in the native soils left in place. The theory is once the primary source of metals is removed and the remaining soil is revegetated, metals will not be available to be mobilized to impact the surface or groundwater. Also, no soil replacement is done. An important aspect in applying to removal technique to for the tailings to be laid into the floodplain in a discrete and visually discernable layer. This is important because determination of where tailings are to remove can be done using visual criteria, instead of chemical analysis.

The construction of Project II requires excavating mine tailings from non-vegetated areas within a section of Silver Bow Creek east of Opportunity extending approximately 2000 feet. The excavated tailings shall be transported to a off-site repository area near Opportunity. The relocated tailings in Opportunity Ponds will be covered with crushed lime rock at the end of the project.

The excavated areas will be graded to drain to Silver Bow Creek and then amended with lime kiln dust to reduce the mobility of any residual metals present in the soils. Some stream banks within the project area will be folded back to a 3H:1V slope. Other banks will be left in their natural condition to provide fish habitat.

The primary advantage of the Project II design is its ability to remediate only the tailings interspersed in unimpacted riparian area. This "surgical removal" of floodplain tailings retains as much of the existing natural setting, thus allowing quicker recovery of the entire area. Also, enmass removal of the fluvial tailings is perceived as the remediation option providing the highest protection to the environment.

These two projects have been designed and will be implemented this year. It is expected these designs will play a key part in the design of the reclamation of Silver Bow Creek and parts of the Clark Fork River in the next five years. The information from these projects, including cost, implementability, and environmental monitoring data to determine the design's effectiveness will be available for application in other riparian settings impacted by mine tailings.

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

THE CLARK FORK BASIN, MONTANA REGIONAL CONCEPTS FOR MEGA-SITE REMEDIATION

Mike Bishop [1]

ABSTRACT

PURPOSE

The purpose of this presentation is to provide an introduction to a series of presentations on the Clark Fork Basin Superfund Sites at the Billings Symposium and to provide regional management concepts that have been developed to address a site of this magnitude. Information provided will include background on the source of the waste materials, a description of the Superfund sites, definition of the human health and environmental problems posed by the wastes, and regional concepts that have been used to prioritize and remediate this extensive Superfund site. Examples of key regional management documents will be available for review.

BACKGROUND AND HISTORY

The Headwaters of the Clark Fork River are located in southwestern Montana. Within the area known as the upper Clark Fork River Basin, world-class mining and smelting operations existed for a period exceeding one-hundred years. During the course of these operations a variety of extraction, beneficiation, and processing waste streams were released. The historic releases of mining and smelting wastes have resulted in situations where threats exist to human health and the environment.

Four Superfund sites are located within the upper Clark Fork River Basin that collectively involve one of the largest Superfund investigations in the nation. The National Priorities List (NPL) sites located within the Basin include the Silver Bow Creek/Butte Area, Montana Pole, Anaconda Smelter, and Mill Town sites. Of these sites, all but the Montana Pole site involve a variety of mining and smelting wastes that contain a variety of heavy metals and arsenic. The focus of this presentation will be on the mining-smelting wastes that are being addressed under Superfund and the Regional Concepts that have been developed to facilitate remediation of these wastes.

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[1] U.S. Environmental Protection Agency, Federal Building, 301 South Park Street, Helena, Montana 59626

Human Health and Environmental Risks

Potential human health concerns are related to toxicity of arsenic, cadmium lead and mercury. Mining wastes containing these contaminants are found in close proximity to residential areas and the potential exists for human exposure via ingestion and inhalation pathways. Ingestion of contaminated soil is a potentially significant route of exposure for young children who may mouth soiled objects and ingest soil. Inhalation of particulates containing arsenic and cadmium may be a significant health threat since these elements are classified as Group B1-probable human carcinogens. Ingestion of lead is of primary concern for human health in some key ares (such as Butte, Montana) because of its potential to affect the blood-forming and nervous systems of the human body, and cause anemia, learning disabilities, brain damage, and renal dysfunction.

The presence of metals in surface waters has adversely affected aquatic life. Copper and zinc concentrations in Silver Bow Creek exceed chronic water quality criteria. This situation has resulted from dissolved contaminants in ground water and surface water, and contaminated suspended material in surface waters reaching Silver Bow Creek. Fish are not present within the upper reaches of Silver Bow Creek. Historically, Silver Bow Creek once had populations of bull trout and cutthroat trout. The current lack of fish in Silver Bow Creek suggests that the high metals concentrations has had a severe impact on the ability of the creek to sustain a viable fishery. In the vicinity of the Mill-Willow by-pass historic fish kills have occurred every two to three years during periods of intense runoff (carrying contaminants into the stream).

Regional Management Concepts

Out of necessity several advanced management concepts have been developed to manage a project as complex and expansive as the Clark Fork River Basin Superfund sites. A regional Master Plan has been developed that takes each of the Superfund sites and provides a frame work for defining smaller manageable operable units. A prioritization scheme is also provided for dealing with the most severe human health and environmental problems first. Schedules of a number of active projects are also monitored via Master Plan updates. Regional documents have been formulated to coordinate laboratory procedures, data validation protocols, the data base management of vast amounts of data, and quality assurance plans that assure that information will meet the quality for its intended use. Each of these documents will be available for review at the Symposium.

CAN WE IMPROVE SHRUB SEEDLING VIGOR BY MANAGING SEED IMBIBITION?

D.T. Booth[1]

ABSTRACT

The absorption of water during seed imbibition is fundamental to germination and growth. However, the stress of imbibition often reduces seedling vigor. Imbibitional stress is influenced by the initial moisture content of the seed, temperature, and the rate at which water is taken up. Research with agronomic species, supported by seed research with conifer and other species, has demonstrated the benefit of controlling imbibitional stress. The available data provide some general guidelines for shrub revegetation, but specific information is needed. Until data are available for western shrubs, reclamationists might assume that, generally, seeds should have 20 to 30% initial moisture before imbibition, and that soil temperatures at native seed collection sites during seed dispersal might be indicative of desirable imbibition temperatures.

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[1] D.T. Booth, Range Scientist, USDA., Agricultural Research Service, High Plains Grasslands Research Station, 8408 Hildreth Road, Cheyenne, WY 82009

WE NEED SHRUB SEEDLINGS WITH GREATER VIGOR

Returning native shrubs to the post-mining plant community in a cost-efficient manner has remained one of the foremost challenges of the western reclamationist. Yet, society is placing increasing emphasis on shrub restoration (WEQC 1992), therefore, the need to improve shrub restoration technology is becoming more critical.

Seeding shrubs directly into a prepared seedbed is not difficult, and is not expensive when successful. Success depends on good seed vigor and rapid seedling development. In this paper I review factors that threaten the transition from seed to vigorous seedling.

THE SEED

The seed is a package in which the delicate machines of life are transported through time and space. We know packaging (seed formation and maturation) is critical. We are learning that unpacking (seed rehydration and germination) is equally important.

During seed development cells multiply, enlarge, and produce food reserves that are deposited as discrete intracellular bodies of carbohydrate, lipid, protein, and organic phosphate. Inorganic compounds of calcium, magnesium, and potassium accumulate in the seed (Bewley and Black 1978, page 28), and may also accumulate in the seed coverings (Booth 1989). At physiological maturity the seed ceases to synthesize. Fresh weight decreases. The seed dries. Activities associated with life are suspended. The seed and its surrounding structures are dispersed or harvested.

To appreciate the dry seed, one must recognize the role of water in a functioning cell. Cell walls and cell organization - in short, cell structure, is maintained by water. The chemical reactions of life require hydrated enzymes. Water is needed for dissolving and transporting metabolic reactants, and as a reactant (as in the hydrolytic digestion of stored food). Dry seeds (8% moisture) have insufficient water to maintain cell structure; there is no measurable respiration, or enzyme activity (Crowe and Crowe 1986, Leopold and Vertucci 1989, Vertucci 1989a).

It is characteristic of maturing seeds to accumulate sugars - sucrose and the raffinose oligosaccharides (Amuti and Pollard 1977, Koster and Leopold 1988). As seed tissues dry and cellular water is removed, the accumulating sugar molecules are believed to replace water molecules around cell membranes and macromolecules (Crowe and Crowe 1986, Leopold and Vertucci 1986, Caffrey et al. 1988). This replacement helps conserve structural form and stabilize the phospholipid bilayers of membranes (Crowe et al. 1989). It prevents damage that would result from changes in relative positions of membrane constituents. In other words, sugar molecules form the "packing" that keeps fragile parts in place within the "package". Proteins, known as "late embryogenesis accumulating" (lea) or "maturation" proteins are also correlated with desiccation tolerance [in soybeans (Glycine max L.)] (Blackman et al. 1991). Their specific function is not known.

IMBIBITION

Imbibition, which is water uptake by the seed, is a period of stress, a time when seedling performance (vigor) is influenced for better or worse. Three interactive factors control seed stress during imbibition: (1) the initial moisture content of the seed, (2) the rate at which water is taken up, and (3) seed temperature (Pollock 1969, Bewley and Black 1978, Vertucci 1989b).

Seed Moisture

Pollock and co-workers established that initial seed moisture affected seedling vigor in lima beans (Phaseolus lunatus L.) (Pollock and Toole 1966, Pollock 1969) and that imbibitional injury was reduced when preimbibition moisture was slowly increased from about 8% to about 20% water (dry wt. basis) (Pollock et al. 1969). Similarly, soybeans imbibed at 13 or 16% moisture produced more seedlings (with a greater dry weight) than when seeds were imbibed at 5 or 6% moisture (Obendorf and Hobbs 1970, Hobbs and Obendorf 1972).

Electron microscope studies of membranes in soybean (Chabot and Leopold 1982) and lettuce (Lactuca sativa L.) (Smith 1991) seeds suggest why initial seed moisture influences seedling vigor. The plasmalemma of the dry seed was described as irregular, and interrupted by pock marks (which might have been salt crystals or lipid inclusions), and had few imbedded particles. Like a machine packed for shipping, the membrane was present, but not entirely functional. In contrast, plasmalemma membranes at 18-20% moisture had corrugated areas interspersed with plaques of smooth conformation (Chabot and Leopold 1982). When fully imbibed the membranes were smooth planes with abundant embedded particles of different shapes and dimensions. Since imbibitional damage occurs during the initial contact of dry seed tissue with water (Leopold and Vertucci 1986), it is possible that the sugar "packing" produces a structural brittleness which makes membrane parts susceptible to the movement and pressure of imbibition. Wetting sugar-packed membranes may lead to displaced membrane parts and deleterious mixing or loss of cellular components and compounds. Yet, at about 20% seed moisture, water molecules have replaced a significant proportion of the sugar packing so that wetting damage is minimized.

Seed moisture also influences cell metabolism. Eight to 25% moisture reactivates the cytosol, glycolytic enzymes, and glycolysis (Wilson and Harris 1966, Ching 1972); 20 to 32% moisture produces expanding metabolism including early mitochondrial function (Leopold and Vertucci 1989).

Respiration begins before seeds are fully hydrated. Under some conditions this is a benefit and a means of promoting rapid germination. It can also reduce stored food, cause an accumulation of harmful oxidation products, accelerate seed aging, and reduce desiccation tolerance (Vertucci and Leopold 1987, Koster and Leopold 1988). The management opportunity is to raise seed moisture levels so membranes and other cell structures are ready for imbibition, but to control the stresses caused by increasing seed respiration too early or too rapidly.

Imbibition Rate

Imbibing seeds have wetting fronts. Cells that are rehydrating are distinct from cells in unwetted areas (Bewley and Black 1978, Vertucci and Leopold 1983). During rapid water uptake, seeds swell and may double in size. Swelling causes pressure inside the seed. Differential swelling at wetting fronts cause mechanical stress. In several legumes (Phaseolus, Pisum, Glycine, Cicer, and Vicia) imbibition has been found to cause cotyledon damage (cracking and breaking), and pressure-driven extrusion of intracellular substances (Ishida et al. 1988, Spaeth 1989). These stresses reduce seedling survival (Duke et al. 1986).

The external morphology of a seed or diaspore can influence or often control imbibition. Hairs, bur spines, pappi, awns - if thoroughly embedded in the soil - or mucilage improve hydraulic conductivity by increasing the absorbing surface area and by enhancing the immediacy of absorbing surfaces with the water-supplying matrix. Conversely, seed appendages not thoroughly embedded in the soil reduce hydraulic conductivity, slow imbibition and therefore buffer the rate of water uptake. The seed coat also influences imbibition, with a greater uptake rate often occurring through the micropyle than through the rest of the testa (Bewley and Black 1976, page 115). Many species have hard seed coats or other chemical/structural characteristics that influence imbibition (Woodstock 1988).

The chemical composition of a seed influences the amount of ‘bound water’ (molecular water) that remains in the "dry" seed. Therefore it influences both the hydrophilic nature (wettability of the tissue) and the amount of water that can be absorbed. There appears to be little relationship between the water potential (the amount of water that can be absorbed) of a seed or seed part and its rate of water uptake. Instead, tissue permeability and wettability largely determine uptake rate (Vertucci and Leopold 1983, Vertucci 1989b).

Temperature

In general seeds absorb moisture more slowly at lower temperatures (Shull 1920, Allerup 1958). One cause is the increasing viscosity (higher surface tension) of water with decreasing temperature, another is the effect of temperature on seed membranes (Murphy and Noland 1982, Vertucci and

Leopold 1983, Vertucci 1989b).

Low imbibition temperature is known to reduce seedling vigor in lima beans (Phaseolus lunatus L.)(Woodstock and Pollock, 1965), garden beans (Phaseolus vulgaris L.)(Pollock et al. 1969) cucumber (Cucumis R.) (Segeta and Tronickova 1966, as cited by Pollock 1969), and peas (Pisum sativa L.)(Highkin and Lang 1966). In contrast, warm (>15°C) imbibition will reduce seedling vigor of winterfat [Eurotia lanata (Pursh) Moq.; Ceratoides lanata (Pursh) J.T. Howell] (Figure 1, Booth 1992). Similar tests with other chenopods and with agronomic species suggest seeds have a specific optimum imbibition temperature (Table 1).

APPLICATION TO MINED LAND REVEGETATION

Most of the work reviewed used agronomic species such as soybeans. Research with woody species show the same principles apply. Controlled rehydration (osmotic conditioning / invigoration regimes) of conifer seeds confirm that managing imbibition can enhance seed vigor (Huang 1989, Huang and Zou 1989, Bourgeois and Malek 1991). Jones et al. (1991) documented reduced germination capacity in noble fir (Abies procera Rehd.) when seeds were soaked at 4°C for 48 hours (a standard practice for prechilling). The authors inferred that soaking the seed (uncontrolled imbibition) caused imbibitional damage. These studies strengthen the premise that careful rehydration of dry seed tissues is important for all species.

When planning seed rehydration, three facets of imbibition temperature should be considered with initial seed moisture. First and second are the effects of warm temperatures on imbibition rate, and on the respiration rate of partially hydrated non-dormant seeds. The third consideration is the specific optimum imbibition temperature. This is mostly unknown. I speculate that seeds which readily absorb moisture have an optimum imbibition temperature corresponding to the soil temperature at the native site during dispersal. Thus, those species like winterfat that disperse seed or diaspores during the late fall or winter can be expected to have a positive response to lower imbibition temperatures. Species like hopsage (Grayia), which disperse earlier in the season, may require warmer temperatures (Table 1). In summary, we know imbibition temperature affects seedling vigor; now we need to know why and how.

SUMMARY AND CONCLUSIONS

Western reclamationists need inexpensive, but dependable methods for restoring shrubs to mined lands. Direct seedings could be more effective if seedling survival could be improved. Advances in seed physiology, particularly in the area of seed rehydration, suggest that managing seed imbibition can enhance the transition from seed to seedling. The majority of research concerns agronomic species with little data for specific application to shrubs. Research on

Figure 1. Influence of imbibition temperature on post-germination growth (axil length) of three winterfat ecotypes (Booth 1992).

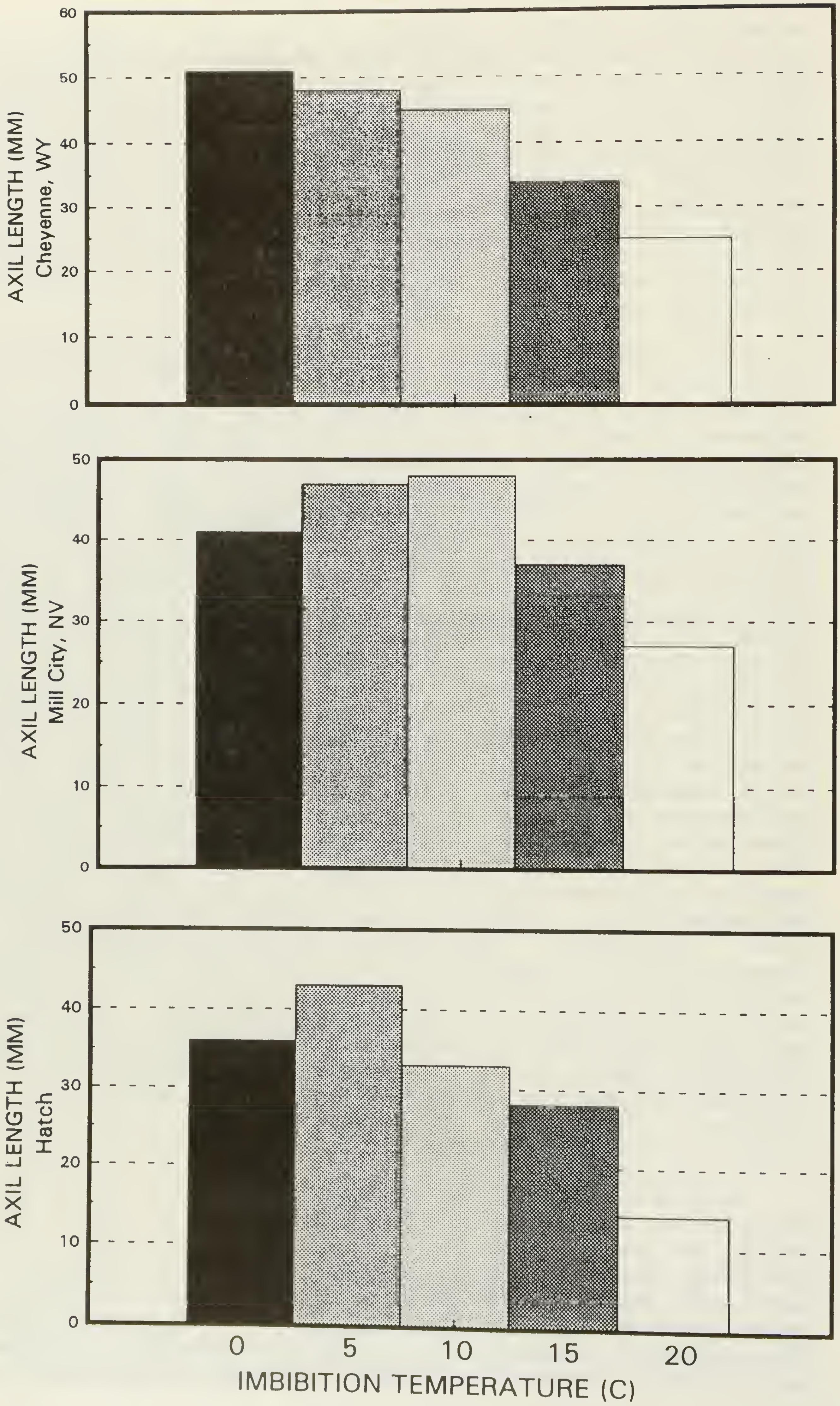

Table 1. Optimum imbibition temperatures as determined by the post-germination growth (axil length) for six species[1].

SPECIES	TEMPERATURE (°C)
WINTERFAT (*Eurotia*)	5
PROSTRATE KOCHIA (*Kochia*)	10
SPINY HOPSAGE (*Grayia*)	15
GARDEN BEAN (*Phaseolus*)	20
TOMATO (*Lycopersicon*)	25
CUCUMBER (*Cucumis*)	30

1. Seed moisture at imbibition was about 20%. Data for winterfat are an average of three replications among three ecotypes (Booth 1992). Data for other species are from five replicated tests of one seedlot (Booth, unpublished data). Significant ($P \leq 0.05$) F-tests for regression analysis or for analysis of variance were obtained for all species. Note that one seedlot may not be characteristic of the species.

shrubs is needed to define optimum imbibition conditions and, to determine if seed treatments to regulate imbibition can economically improve seedling vigor. Until specific information is developed, reclamationists might assume that dry seeds should have 20 to 30% moisture before imbibition, and that imbibition temperatures should be near those found at native seed collection sites during seed dispersal.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

USE OF PRESCRIBED BURNING IN REGENERATING WOODY DRAWS

Carolyn Hull Sieg[1]

ABSTRACT

Native deciduous woodlands, or "woody draws," constitute less than 1% of the land area in the Northern Great Plains, but provide critical wildlife habitat and contribute greatly to the biotic diversity of this region. Unfortunately, research has shown that many of these woodlands are in degraded condition, characterized by little or no shrub reproduction, few or no tall shrubs, and an understory of mostly Kentucky bluegrass (Poa pratensis). The objective of this study was to test the feasibility of using prescribed burning to rejuvenate two types of woody draws: 1) bur oak (Quercus macrocarpa)/chokecherry (Prunus virginiana) stands in the foothills of the Black Hills and 2) green ash (Fraxinus pennsylvanica)/chokecherry woodlands in Badlands National Park. Both studies indicated that prescribed burning shows some promise in regenerating decadent woody draws. Burning increased sprouting rates of green ash and bur oak relative to unburned sites and in some cases reduced the cover of Kentucky bluegrass. These results also indicate that conditions necessary for burning stands with sufficient intensity to stimulate shrub and tree sprouting are most likely to occur in the fall in this region.

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[1] Rocky Mountain Forest and Range Experiment Station, South Dakota School of Mines Campus, 501 E. St. Joseph, Rapid City, SD 57701.

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

MYCORRHIZAE AND MINED LAND RESTORATION: CURRENT STATUS AND FUTURE DIRECTIONS

Helen Atthowe[1]

ABSTRACT

Mycorrhizae enhance the field performance of native trees and shrubs through capture and uptake of nutrients; buffering against moisture stress, root pathogens, and toxic levels of mineral elements; and improving soil structure. The potential contribution of mycorrhizae to the successful revegetation of mined lands is discussed.

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[1] Plant Production Manager, Bitterroot Native Growers, Inc., 455 Quast Lane, Corvallis, Montana, 59828.

INTRODUCTION

Most vascular plants in undisturbed soils form mutually advantageous relationships between their roots and soil microorganisms. The most common plant-microbe relationship is the symbiosis between certain non-pathenogenic fungi and plant root cells, called a mycorrhizal association. Mycorrhizae literally means "fungus-root". Both the fungus and the host plant derive benefit from the association. The fungus gains a place to live and a source of carbohydrate supplied by plant photosynthesis. The host plant benefits, because fungal hyphae effectively extend the plant's root absorbing surface area throughout the soil, beyond zones of nutrient and water depletion.

There are two major groups of mycorrhizae, differentiated by the way in which they infect plant roots. Ectomycorrhizae (EM) grow around the exterior of the plant root tip, with fungal hyphae extending into the soil. EM occur primarily on members of the Pinaceae, Betulaceae, and Fagaceae. Endomycorrhizae (ENM) grow inside root cells and cannot be seen with the naked eye. Like EM, ENM send hyphae out from the plant root into the soil. A special type of ENM form structures inside plant roots called arbuscules and vesicles. This type of ENM is called vesicular-arbuscular mycorrhizae (VAM). VAM are common in 90% of the higher plant groups on earth, including many crop plants and many of the native trees and woody shrubs used in reclamation. A third mycorrhizae group is called ectendomycorhizae. It combines features of both EM and ENM, and is found on Ericaceae species. Within each of the three groups of mycorrhizal fungi, there are many different genera and species of fungi.

BENEFITS OF MYCORRHIZAE

Mycorrhizae associations have been known for over 100 years, but it is only in the past 35 years that their contribution to plant growth has been appreciated. It is generally accepted that mycorrhizae enhance survival and growth, especially on harsh sites. ENM reportedly improved growth and survival of Four Wing Saltbush (*Atriplex canescens*) and other woody shrub species on coal mine spoils in the arid west (Aldon 1975, 1978, 1980). VAM enhanced the growth of native woody plants used for phosphate mine reclamation (Sylvia 1990). Both EM and ENM inoculated Live oak (*Quercus virginiana*) and Chinese Tallow Tree (*Sapium sebiferum*) showed greater survival and growth in lignite overburden in Texas (Davies 1990). EM enhanced the growth of Pinus species on striped mined bauxitic soils in Jamaica (Howard 1991). VAM have reportedly increased plant height, stem diameter, biomass and growth rates of woody shrub species (Danielson and Visser 1989, Vinayak and Baryaraj 1990, Visser et. al. 1991). There are cases, however, in which mycorrhizal inoculation did not enhance seedling survival (Pilz and Znerold 1986). Seedling quality may be involved when mycorrhizae inoculation does not exhibit a positive growth response. Barnett

(1984) reported that larger non-mycorrhizal container-grown seedlings (grown under a high fertilizer program) survived better than mycorrhizal seedlings inoculated early in their development and grown under a low fertility program. Linderman (1987) suggested that if seedlings do not have healthy, well-established root systems, mycorrhizal infection is significantly reduced.

One reason mycorrhizae effect plant growth positively is their enhancement of mineral nutrient acquisition, especially the uptake of P, Cu, and Zn (Mosse 1986, Kimmins 1987, Linderman 1988b, Vinyak and Bagyara 1990). Mycorrhizal associations can alleviate P deficiency problems on high P-fixing soils (Linderman 1988). Mycorrhizal hyphae release mineral nutrients into the soluble forms required by most plants, when the organic acids they secrete attack undecomposed soil minerals and organic matter (Kimmins 1987). This fact is especially important in high Ca western soils that fix P into insoluble $CaPO_4$ complexes. The ability to take up P rapidly imparts a high level of competitiveness to mycorrhizal plants, since P is essential for initial root development and early plant growth.

There appears to be a correlation between mycorrhizae and nutrient distribution in plant organs. Miller and Rudolf (1986) reported a greater concentration of Mn in inoculated Lodgepole pine (*Pinus contorta*) shoot tissue. Mn accumulated in the root tissue of non-inoculated pines.

Mycorrhizae may also control mineral uptake in another way. Mycorrhizae effect plant membrane permeability and act as a buffer against excessive uptake. Mn toxicity is alleviated by VAM in soybean (Bethlenfalvay et al. 1989). Mycorrhizae reportedly protect plants against Cu and Zn toxicity (Berry and Marx 1976, Bradley et al. 1981). In fact, mycorrhizae have been shown to reduce heavy metal content in shoot tissue (Bradley et al. 1981) and modify plant roots such that they are better able to tolerate mineral toxicity (Cordell et al. 1991). Kasuya et al. (1990) reported that mycorrhizae ameliorated Al damage to plant roots. In other cases, mycorrhizae have improved plant tolerance to saline soils (Ojala 1982, Linderman 1988b). Besides helping plants to cope with low fertility soils, mycorrhizae may also help plants tolerate low Ph (as low as 4.0) and high soil temperatures (Cordell et al. 1991). Mycorrhizae enhance plant nutrition in one final way. They appear to stimulate other soil microorganisms, including P-solubilizing bacteria, N-fixing bacteria, and plant growth promoting rhizobacteria (Linderman 1988a).

Soil drought is one of the most important environmental stresses mycorrhizae mitigate. The mechanism is not yet agreed upon, but mycorrhizae induced plant drought tolerance seems to be related to the increased P uptake afforded by mycorrhizae. As already mentioned, Mycorrhizae extend plant root systems beyond zones of water depletion. There are many reports of mycorrhizal plants withstanding soil drought better than non-mycorrhizal plants (Marx et al.

1984, Allen and Allen 1984, 1986, Kimmins 1987, Nelson 1987, Ibrahim et al. 1990, St. John 1990).

Mycorrhizal association results in plant primary and secondary metabolism alterations (Chalot et al. 1989). In fact, VAM associations have been correlated with higher rates of photosynthesis (Linderman 1988b). Mycorrhizae reportedly reduce transplant shock (Bierman and Linderman 1983) and enhance rooting of cuttings (Stein et al. 1990). Mycorrhizae may influence plant metabolism directly as well. Slankis (1973) reported that some mycorrhizal fungi produce plant growth promoting substances such as auxins, cytokinins, giberellins, and growth regulating B-vitamins.

Mycorrhizal plants are reportedly less susceptible to certain diseases than are non-mycorrhizal plants (Dehne 1982, Linderman 1988b). Sinclair et al. (1975) demonstrated that EM suppressed *Fusarium* in a nursery situation. Mycorrhizae reduced disease by producing antibiotics, influencing sporulation patterns of pathogens, and competing for infection sites (Dehne 1982). Mycorrhizae may also trigger phytoalexin production and enhance defense mechanisms, such as chitinase (Linderman, personal communication).

The benefits of mycorrhizae extend beyond the host plant to the entire plant community, including the soils into which mycorrhizal plants are transplanted. Mycorrhizal hyphae have been shown to distribute recently-fixed Carbon throughout the soil providing energy for the entire microbial community (Norton et al. 1990). Mycorrhizal hyphae extend several meters into the soil, providing a source of high carbon organic matter to the system. In fact, mycorrhizae have been shown to increase soil organic matter contents (Mosse 1986). Increased soil organic matter content usually results in more stable nutrient cycling and improved waterholding capacity of droughty soils. Mycorrhizae also positively effect soil aggregation and thus soil stability. The mycelia of VAM fungi are coated with a mucilaginous substance that causes soil particles to adhere. More soil aggregates were found in soils with mycorrhizal plants (Tisdall and Oades 1979). Finally there are suggestions that mycorrhizal plants transfer nutrients and carbon to other plants by way of a hyphal network. Goodwin (1992) reports that VAM transfer nutrient assimilates from established plants to seedlings.

MYCORRHIZAL CONTRIBUTION TO IMPROVED REVEGETATION SUCCESS

Mycorrhizal inoculation of seedlings transplanted onto disturbed sites may improve survival and growth in one or more of the following ways: Mycorrhizae enhance a plant's ability to acquire nutrients and water on sites where one or both are limited. In fact, the benefits of VAM are reported to be greatest in nutrient poor soils and extreme environments (Bethlenfalway 1984, Allen and Allen 1986). Mycorrhizae alleviate transplant shock under the dry and hot conditions most revegetation project plants face. Mycorrhizae provide plant growth promoting and regulating substances, and improve plant resistance to root disease. Mycorrhizae increase photosynthetic rates and therefore promote faster plant establishment and earlier maturity. On difficult reclamation sites, mycorrhizae can alleviate excess mineral elements and chelate metal ions such that plants are buffered from their toxic effects. Mycorrhizae improve soil structure and nutrient cycling capacity, thus enhancing growth and reestablishment of an entire plant community.

Mycorrhizae are sensitive to soil disturbance (Reeves et al. 1979, Birch et al. 1991) and are significantly decreased in number by mineral extraction activities (Allen and Allen 1980, Birch et al. 1991, Visser et al. 1991). The rate of spread of VAM back into reclaimed soils after disturbance is very slow (Johnson and Macgraw 1988 a and b) since VAM spores are large and not easily carried by the wind (St. John 1990). Mott and Zuberer (1987) reported that VAM took 3-7 years to return to pre-mining levels after reclamation and revegetation. Unfortunately, when mycorrhizal fungi recolonize a site slowly, plants that depend on mycorrhizal associations are less able to compete with non-mycorrhizal plant species. The non-mycorrhizal species that may colonize reclaimed sites initially are usually ruderal or weedy species such as Russian Thistle (*Salsola kali*) and Kochia (*Kochia scoparia*) (Allen and Allen 1980).

Since restoration of a stable plant community is the goal, most plant species used in revegetation are later successional species. In general, late successional plant communities include more mycorrhizal species than do early successional communities (St. John 1990). Wilson et al. (1991) suggested that mycorrhizae are an integral component in site recovery from disturbance. St. John (1990) cautions that, though the mycorrhizal condition of some of the native plants used in restoration is unknown, mycorrhizal inoculation is a prudent precaution. Goodwin (1992) suggested that VAM inoculation may play an important role in the restoration of rangeland communities.

BITTERROOT NATIVE GROWERS MYCORRHIZAL INOCULATION PROGRAM

At Bitterroot Native Growers, we are convinced that the benefits of mycorrhizae warrant the time we have spent (and continue to spend) to develop an inoculation program. In general, mycorrhizae do not occur in significant numbers naturally in the nursery. In a survey of commercial nurseries, Danielson and Visser (1990) reported that eight species of container-grown woody shrubs were rarely VAM if less than one year old, and erratically VAM after one to two years in the nursery. Pinus species surveyed ranged from EM-free to completely EM-colonized. In order to ensure some predictability, our approach has been to inoculate all conifer species with commercially cultured Rhizopogon EM. We are also inoculating deciduous species (such as Oak) and woody shrub species (such as Sage) with EM and ENM that we culture at the nursery ourselves.

We have developed a unique systems approach to EM and ENM inoculation that fits into our production program. There are several points our program considers. First, although many researchers consider that mycorrhizae, especially VAM, are not host specific, there is evidence that some mycorrhizal fungi form plant species-specific associations (Giovannetti and Lioi 1990) and plant cultivar-specific associations (Linderman and Call 1977, Chavez et al. 1990). Since we are often unsure which type(s) of mycorrhizal fungi are best for particular plant species and particular revegetation situations, we have adopted a program that encourages both EM and ENM growth. We collect soil and root pieces from healthy, native stands of the plant species we plan to grow. We then culture whatever types of mycorrhizae are present, using pot culture. This approach also allows us to bring along other soil microorganisms that may be present. This is important since we subscribe to Linderman's "microrhizosphere" theory. Linderman (1988a) suggested that there are significant interactions between mycorrhizae and other soil microorganisms and that the reported effects of VAM in natural soils may actually be the combined effects of VAM and microbial associates. Chanway and Holl (1991) supported this theory by showing that Lodgepole Pine (*Pinus contorta*) growth was stimulated to a greater extent when coinoculated with both mycorrhizae fungi and a growth promoting bacillus stain, than when inoculated with either separately.

The final point our approach takes into consideration is the relationship between mycorrhizae and P and N fertilization. In general, mycorrhizae are negatively affected by high levels of N and P fertilizers (Mosse 1986). However, Davis (1990) reported that both EM and ENM fungi enhanced plant growth in N-deficient lignite mine reclamation sites, on both high and low P-level soils. We inoculate after our plants have passed through their exponential growth phases, in order to keep fertilizer rates high enough to assure high quality plant development. We decrease fertilizer P and N levels significantly after inoculation.

FUTURE RESEARCH DIRECTIONS

One important area that needs more study is the development of soil disturbance and storage techniques that attempt to maintain native mycorrhizal fungi populations. Another area of research needed is a comparison of plant growth, when inoculated with native verses introduced (or exotic) mycorrhizae. One study reports greater growth responses using introduced *Pisolithus-Tinctorius* (Pt) EM fungi as compared with native EM fungi (Berry and Marx 1976). However in another study comparing 9 strains of EM fungi introduced onto Jack Pine (Pinus banksiana) seedlings grown in oil-sands tailings, it was reported that after three years only 1 out of 9 fungi tested was present in significant numbers, and 72% of the transplants' roots were converted to indigenous mycorrhizae (Danielson and Vissor 1989). A final research goal should be continued study of interactions between other soil microbes and mycorrhizae, so that some day we might be able to inoculate with the optimal combinations of microbes for each revegetation project.

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Planning, Rehabilitation and Treatment of Disturbed Lands
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UTILIZATION OF SITE-ADAPTED CONTAINERIZED PLANT MATERIAL FOR SUCCESSFUL WOODY PLANT RESTORATION

Vern Pfannenstiel[1], Leonard Ballek[2] and Jan Krueger[3]

ABSTRACT

The Seneca II surface coal mine is located in northwestern Colorado on the northern flank of the Williams Fork Mountains. The rolling topography is dominated by mountain shrub and sagebrush grassland vegetation that provides important habitat for a variety of wildlife species, particularly big game. Reestablishment of shrubs in reclaimed areas has been an important and difficult reclamation objective. Efforts to achieve this objective have included the planting of containerized and bare root seedlings. Utilization of non-adapted planting stock, inadequate site selection and preparation and poor planting procedures have contributed to poor seedling establishment and survival. In 1988, efforts to improve the woody plant establishment program included contracting with Bitterroot Native Growers (BNG) to supply and outplant containerized trees and shrubs. In 1989 and 1990, BNG and Seneca II Mine personnel worked to refine site preparation, species site placement and planting techniques. In 1990, shrub seed was collected from the mine site to develop site-adapted containerized planting stock for two principal tall shrub species. Plant survival has increased dramatically from 1988 to 1991. The use of site-adapted planting stock and improved planting and cultural practices have contributed to improved establishment of woody plants at the Seneca II Mine. Ongoing work, including seed collection for additional species and mycorrhizal inoculation of seedlings, should continue to improve woody plant reestablishment.

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[1] Senior Environmental Scientist, Peabody Coal Company, 1300 South Yale, Flagstaff, Arizona 86001

[2] Operations Manager, Bitterroot Native Growers, P.O. Box 566, Hamilton, Montana 59840

[3] Vice President, Bitterroot Native Growers, P.O. Box 566, Hamilton, Montana 59840

INTRODUCTION

The Federal Surface Mine Control and Reclamation Act of 1977 (P.L. 95-87) and Colorado's similar act require environmentally sound coal mining operations and appropriate reclamation of disturbed lands. Implementation of the Federal and State act is achieved under Colorado's approved permanent regulatory program. These regulations, in part, state that a diverse, effective, and permanent vegetative cover be established which is native to the area and is capable of supporting the approved postmining land uses. In northwestern Colorado, mining occurs on native rangeland used for livestock grazing and as wildlife habitat. In most cases, similar uses are designated for the postmine environment. The postmine land uses, in turn, dictate the effort and technologies necessary to successfully complete reclamation. Where wildlife is a designated postmine land use, a return of woody plants on the reclaimed landscape is required. Further requirements for woody plants include planting arrangements, minimum densities, and utility for wildlife species. The requirement for a return of woody plant density has been one of the more controversial and difficult aspects of successful reclamation.

Major shrub dominated communities in northwest Colorado include the mountain brush and sagebrush grassland types. These regionally extensive native plant communities are characterized by high densities of woody plants. At Peabody Western's (Peabody) Seneca II Mine, typical mountain brush stands have over 10,000 stems per acre, while a sagebrush grassland community may average 6,000 stems per acre. Livestock grazing in the region for over 100 years has likely contributed to an increase in woody plant density due to removal or declining vigor of herbaceous vegetation and a subsequent competitive advantage to woody vegetation. Increased woody plant density under grazing pressure is common for shrublands in the west (Valentine, 1971).

Regulations require woody plant density either be statistically equal to 90 percent of the premine or reference area density, or a lesser density if compatible with the postmine land use. The current approved density standard for the Seneca II Mine is 1,000 stems/acre. Although this performance standard is a significant reduction from premine densities, it remains a difficult goal to attain. The reasons for this are more complex than simply disturbances to vegetation and soils. Consideration must be given to the ecology of native shrublands and ecophysiology of native shrubs, changes in soil and substrate characteristics, changes in soil microbiota, competitive interactions between woody plants and herbaceous species, and successional dynamics in native and reclaimed lands that influence timeframes, patterns and pathways of succession (Wali, 1980; DePuit, 1986; Mclendon and Redente, 1990; Neilson and Wullstein, 1986). The removal of sheltering vegetation exposes reclaimed sites to drying winds and direct sun. As a result, disturbed soils may be more prone to higher evapotranspiration rates than native undisturbed soils. When topsoil is removed prior to mining then replaced on recontoured sites, soil structure, organic matter content and soil microorganisms may be adversely affected (Allen and Allen, 1980; Birch et al., 1991).

Appropriate technologies are used to meet difficult reclamation objectives. For reestablishment of woody plants these include selection of site-adapted plants, use of quality nursery stock or seed, reestablishment of mycorrhizal relationships, proper site selection and preparation, proper out-planting procedures, reduction in herbaceous competition, and management of woody plant establishment areas (Wyoming Game and Fish Department, 1981; Allen, 1984; Monsen, 1990). Methods to establish woody plants on reclaimed lands have included planting of bare root and containerized seedlings, direct seeding of shrubs, maximum use of direct-haul topsoil, and live shrub clump transplants (Crofts and Parkins, 1979; Howard and Samuel, 1979; Pfannenstiel and Wendt, 1984; Kiger et al., 1987; Harding and Agnew, 1990).

A combination of the above technologies and procedures have been applied at the Seneca II Mine since 1979. Results have been variable. Analysis of reclamation practices and monitoring of woody plant establishment areas has resulted in a continuing evolvement and improvement of the woody plant reestablishment program. Preliminary results from the improved application of woody plant establishment technologies at the Seneca II Mine indicate improved establishment and survival for containerized seedlings.

Site Description

The Seneca II Mine is located on the northern flank of the Williams Fork Mountains in northwest Colorado approximately nine miles southeast of the town of Hayden. The mine topography is characterized by steep, rolling hills. Elevations range from 6200 feet in Grassy Creek to 8100 feet at Chavez Peak. Slopes range from 2 percent greater than to 40 percent with the majority over 15 percent. Two coal seams are recovered, the Wadge and the Wolf Creek. The 10-feet thick Wadge seam dips at 11 to 34 percent, while the 15-feet thick Wolf Creek seam dips 12 to 25 percent. These conditions create challenging mining and reclamation situations.

Soils on the mine site are derived from interbedded shale and fine-grained sandstone sediments. The soils are generally loamy in nature with a high organic matter content. Finer textured soils with high clay content commonly occur near the Wadge coal recovery line. Soil depths range from less than one-half foot near ridge crests to over five feet in concave swales and drainages.

The climate in the area is semi-arid steppe characterized by cold winters and moderately warm summers. Temperatures range from -20 degrees to 95 degrees Fahrenheit. The annual precipitation is approximately 18 inches with about one-third falling as snow.

The dominant vegetation on the site is the mountain brush type. Principal species in this type include Gambel oak (Quercus gambelii), serviceberry (Amelanchier alnifolia), and chokecherry (Prunus virginiana). The sagebrush grassland type is less extensive but commonly occurs throughout the mine area. Visually dominant species include big sagebrush (Artemisia tridentata) and mountain snowberry (Symphoricarpos oreophilus). Scattered aspen (Populus tremuloides) occur in the more mesic sites of the mine area.

Previous Woody Plant Establishment Results

Greater than 90 percent of the woody plant composition in reclaimed areas is contributed by big sagebrush and snowberry with only a small percentage contributed by tall shrub species. Although the maximized use of direct haul topsoil has resulted in the greatest contribution to reclaimed woody plant density, it has also caused the density to be comprised primarily of sagebrush and snowberry. The Colorado Division of Minerals and Geology and the Colorado Division of Wildlife have expressed concern over the lack of tall shrub presence in reclaimed area woody plant composition. The direct seeding of shrubs has not proven effective. The planting of over 60,000 containerized and bare root seedlings since 1979 has improved establishment of tall shrub species to a limited extent.

The effort to establish tall shrub species has primarily centered on the use of containerized and secondarily, bare root nursery stock. A variety of species normally occurring in northwest Colorado have been planted. The majority of planted species include chokecherry, serviceberry, Gambel oak, and aspen. Prior to 1988, planting was conducted by mine personnel and volunteer help. Site selection and preparation was not optimized. Problems in application and inherent difficulties in reclamation have resulted in low success. Sites planted prior to 1988 have had poor survival. Plantings in a 1986 site concentrated on regional native planting stock, site preparation, planting techniques, and handling of planting stock. In a 1989 survey of this site, survival for chokecherry was estimated to be 34 percent, while serviceberry was 2 percent and Gambel oak less than 1 percent. A visit to the site in 1991 indicated the same survival levels, but vigor and growth of surviving plants was poor. In a 1988 shrub establishment plot, survival transects sampled in 1989 indicated a survival rate for chokecherry of 30 percent, serviceberry 20 percent and aspen 20 percent. Poor site selection, continuation of the dry conditions experienced in 1987 and wildlife depredation contributed to the poor performance in the 1988 plot.

Despite these difficulties, it was decided the program to reestablish woody plants, and in particular tall shrubs, would continue to be based primarily on the use of containerized planting stock. In 1988, an analysis of the planting program indicated that any reasonable success could only be achieved by improving all aspects of the planting program and using site adapted nursery stock. Since 1988, the program has concentrated on a combined ecological and technological approach using Peabody resources and professional outside services.

METHODS AND MATERIALS

Beginning in 1988, Peabody contracted with Bitterroot Native Growers (BNG) to provide plant material and outplanting services at the Seneca II Mine. A number of propagation and outplanting techniques have been used by BNG to improve the survival rates of woody plants at the mine site. These techniques include: using plants native to the site, custom seed collection to assure genetically adapted stock, the use of container grown seedlings, the planting of dormant seedlings, fall planting, use of browse repellents, careful handling of planting stock, and attention to outplanting methods. Peabody has worked on improved site selection and preparation.

Use of Native Plants and the Production of Genetically Adapted Stock

The first planting of BNG stock at the Seneca II Mine in 1988 consisted of chokecherry, serviceberry, and quaking aspen. Gambel oak was added in 1992. These species represent the dominant tall shrub species in the native plant communities at the site. Seed was collected on site in 1990 for chokecherry and serviceberry and in 1991 for Gambel oak in order to develop genetically adapted planting stock.

The use of genetically adapted native plant material is critical for successful long-term survival and reproduction. Plant populations that evolve in a particular environment are adapted to the extremes that periodically occur, such as drought periods or early severe frosts. Critical survival traits determined by the genetic make-up of a population include: susceptibility to winter damage and moisture stress, bud flush and bud set dates, and flowering and dispersal dates (Cleary et al., 1978). Populations that are adapted to other regions may lack the evolutionary advantages needed for long-term survival and reproduction (Millar et al., 1989). The importance of using genetically adapted stock at the Seneca II Mine may be particularly critical because the site is near the northern range limit for the mountain shrub communities, and in particular for Gambel oak (Neilson and Wullstein, 1986).

Containerized Seedlings

Containerized seedlings are used at the Seneca II Mine because of their advantages over bare root stock. The major advantages of containerized seedlings include their reportedly greater field survival than bare root seedlings, a shorter production cycle, an undisturbed root system, extended planting season, and ease of outplanting (Scheetz, 1978; Cleary et al., 1978).

BNG's container nursery can produce dormant seedlings ready for outplanting in 12 to 16 months from seed collection. This short production time allows for quick response to changes in revegetation needs and accelerates the entire revegetation process.

The containers utilized are primarily ten cubic inch Leach "super cells" that are 1-1/4-inch in diameter and 9 inches deep. They are designed with vertical ridges on the interior. These ridges force the roots straight down and out the bottom of the cell where they are air pruned, producing a multi-branched intact root system. Twenty cubic inch Spencer Lemaire cells are used for Gambel oak.

Containerization extends the planting season beyond bare root nursery availability. This is particularly important on arid or disturbed sites because containerized seedlings can be planted when moisture conditions are optimum, rather than when seedlings are available (Romero et al., 1986). This allows greater operational flexibility in the planting program.

Seedlings grown in containers are much easier to outplant and to handle. Unlike bare root, they do not require special storage facilities and care. The compact root systems are also much less prone to improper planting. Ease of planting may be very important, especially when working with inexperienced planters and on adverse sites.

Dormant Seedlings

All seedlings planted at the Seneca II Mine are planted in a dormant condition. Dormant seedlings experience very little transplant shock, resulting in rapid initial growth and establishment (Cleary et al., 1978). Because the moisture requirements of dormant stock are much lower than those of actively growing plants, planting can be done at a time when soil moisture is less than optimum.

Fall Outplanting

Outplanting at the Seneca II Mine has been conducted exclusively in the fall. Planting operations are restricted to fall because summers at the mine site are dry and late snows or muddy conditions in spring may delay planting. Fall precipitation is usually more dependable than spring and winter moisture is generally good. In the past, soil moisture has generally been adequate in the fall. Dormant trees and shrubs planted in the fall have a period of rapid root growth. This enables the seedlings to establish a root system that will help them survive the critical first year.

Outplanting Techniques

Proper outplanting techniques are as important as the quality of the stock used. Even high quality trees or shrubs, improperly planted, have a very high mortality rate. Plant material handling, microsite selection, removal of competing vegetation, proper planting depth, and good soil to root contact are critical.

The handling of seedlings before planting has been critical, especially if the weather is warm. Containerized seedlings are kept in a cool unheated shed until the day of planting. Boxes transported to the field are opened for ventilation and kept in the shade. Containerized stock does not require any additional preplanting preparation such as the "jelly rolling" required by bare root stock.

The most favorable planting sites are north facing slopes or depressions. Disking of existing vegetation has been used to control herbaceous vegetation when freshly topsoiled sites are not available. Proper selection of planting microsites has been emphasized by BNG. Planting in low spots that gather moisture or in the shade of debris can increase seedling survival and growth (Cleary et al., 1978). At each site, the most favorable conditions for establishment of each species is defined by BNG supervisors. Planters then use this information to select the favorable microsite for each seedling. Special attention is given to the natural patterns of vegetation on nearby undisturbed sites and attempts are made to simulate these patterns in the planting process.

When a planting spot is selected, an 18 x 18 inch area is cleared of organic debris and any existing vegetation. This "scalp" decreases the competition from herbaceous vegetation and forms a depression that collects and holds water.

The seedling is removed from the container just prior to planting and placed with roots fully extended downward in the planting hole. Seedling survival and growth may be reduced if roots are planted in a J or L-shape (Cleary et al., 1978). Each seedling is planted with its original plug buried 1/2 to 1-inch below the soil line, but with none of the foliage buried. The soil is carefully firmed around the plant to ensure proper soil to root contact and to prevent air pockets.

Micorrhizal Inoculation

BNG has developed a unique "systems approach" to mycorrhizal inoculation that fits into the seedling production program. It is often questionable which type(s) of mycorrhizal fungi are best for particular plant species and revegetation situations. Therefore, a program has been developed that encourages both endo- and ecto-mycorrhizae fungi growth.

Soil and root pieces are collected from healthy, native stands of the plant species to be grown. In the case of the Seneca II Mine, these materials were collected from the same plant communities from which seed was collected. After soil collection, the types of mycorrhizae present are cultured in the greenhouse using a pot culture method. This approach allows other soil microorganisms that are present to be cultured with the mycorrhizal fungi. Linderman (1988) suggests there is a significant interaction between mycorrhizae and other soil microorganisms, and the reported effects of vesicular-arbuscular mycorrhizae (VAM) in natural soils may actually be the combined effects of VAM and microbial associates.

Inoculation occurs after the plants have completed their exponential growth phases in order to keep fertilizer rates high enough to assure high quality plant development. BNG is presently monitoring Gambel oak planted at the Seneca II Mine in 1992 to determine the benefits of using mycorrhizae-inoculated seedlings versus mycorrhizae-free seedlings. Future inoculated seedlings will include chokecherry and serviceberry.

RESULTS AND DISCUSSION

Improvements in establishment, survival, and vigor have been observed in plantings since 1989. Qualitative observations of planted sites and estimates from survival transects have been used to gauge improvements. Excavations of two-year-old planted stock have been made to measure root extension and observe rooting patterns.

Survival estimates in 1991 for chokecherry and serviceberry planted in 1990 ranged from 75 to 95 percent. For chokecherry and serviceberry planted in 1991, survival transects sampled in 1992 yielded survival estimates ranging from 70 to 90 percent. This represents a dramatic improvement in survival from plantings prior to 1989. Aspen survival continues to be poor with 1991 plantings estimated to have less than 20 percent survival one year later. A lack of reclaimed sites suitable for aspen is a primary reason for that species' poor success. Survival estimates are preliminary at this time and continued monitoring of plantings will be conducted to determine long-term survival, growth, and development.

In October of 1991 and 1992, seedlings were excavated to observe the degree of root growth. Shrubs that had been planted for one year had an average root mass that was 4 to 6 inches in diameter with a tap root of 20 inches. Two year root systems were 6 to 8 inches in diameter with tap roots that exceeded 26 inches. Tap roots beyond the 26-inch depth could not be excavated. Several shrubs that had been severely browsed were also excavated and their root systems were comparable to those of unbrowsed plants. The rapid establishment of extensive root systems improves a seedling's ability to survive adverse conditions such as drought or browse damage.

Site preparation either by disking or providing unseeded freshly topsoiled planting sites has aided in reducing perennial herbaceous competition through the first several growing seasons. Disking is now carried out several times during the growing season to fallow the site similar to windbreak establishment procedures. Disked sites have less surface compaction and improved planting conditions.

Browse damage has been reduced through the first growing season by application of browse repellents. Deer-Away browse repellent has been used since 1989. Initially, individual seedlings were treated in the field when planted. For the last two years the repellent has been sprayed on the seedlings en masse at the nursery prior to shipping. This has reduced costs and improved field efficiency.

Gambel oak seedlings inoculated with mycorrhizae were planted in fall 1992. A control plot was established with replications of inoculated and non-inoculated Gambel oak seedlings to evaluate results. It is anticipated this practice may improve reestablishment of important soil microbial-plant relationships and enhance growth and development.

Future measures include improved site selection and preparation to aid in moderating moisture regimes. Longer-term herbaceous vegetation control through the use of pre-emergent herbicides will be researched. The rapid increase in herbaceous vegetation creates competition problems, yet measures to provide better control are limited. Browse damage to seedlings increases after the second year and may result in severe losses. Browse control efforts will include investigations into the use of longer lasting systemic repellents (e.g., Ani-pel) and the temporary use of standard or electric game fences until plantings are well established. Mycorrhizal inoculation of seedlings will continue to be developed and results monitored.

The monitoring and documentation process is critical to program development and improvement. Planting sites are now surveyed by mine engineering staff for accurate relocation in the future. Within sites, survival transects will be increased in size and number and will be placed to better represent a range of site and planting characteristics. Survival transects will include permanent stakes at individual plants to facilitate documentation. Standard shrub density quadrats sampled during revegetation monitoring have not proven effective in survival estimates. Difficulties in observing seedlings and quadrat placement in planting areas have resulted in much lower survival estimates when compared with staked permanent transects.

Preliminary results indicate an improvement in the woody plant establishment program at the Seneca II Mine using site adapted containerized planting stock in association with appropriate site and cultural practices. The use of professional nursery and planting services specializing in native plants has been indispensible in achieving these improvements. Continued development of the program will include further refinement of these practices, increased use of mycorrhizal inoculants, and improved monitoring to better document results.

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MATURE SHRUB TRANSPLANT SUCCESS ON RECLAIMED MINE SPOILS IN NORTHWEST COLORADO

by

W.S. Harding Jr.[1]

ABSTRACT

Woody plant re-establishment on semi-arid mine lands has historically been and currently is a reclamation challenge for surface coal operators. Federal (SMCRA, 1977) and State regulations require the re-establishment of woody species for specific postmine land uses, particularly for wildlife habitat.

Vegetative success is measured by adequate vegetative cover, production, diversity and woody species densities. An important factor when addressing vegetative re-establishment issues is State primacy. States have the opportunity to establish success criteria which take into consideration enviro-climatic variations within specific regions that make sound, cost-effective and self-sustaining vegetation a site specific endeavor.

There is a need for exchanging technology concerning woody species re-establishment efforts. Trapper Mine has applied and studied various woody species establishment techniques since the early 1980's. These include direct seeding, seedling planting, direct-haul topsoil re-establishment, and mature shrub transplanting.

Survival studies for mature shrub transplanting at Trapper examine differences in effective moisture, and equipment sizes (front end loaders (FEL)) in relation to overall mature shrub survival. Since 1985, overall clump survival ranges from 65 to 100 percent. Additionally, these shrub clumps contribute over 200 woody stems/acre to the total woody stem density. Costs, FEL efficiency and adequate transplant "pad" sizes are also discussed.

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[1] Environmental Engineer, Trapper Mining Inc., P.O. Box 187, Craig CO 81626.

INTRODUCTION

Shrub re-establishment on reclaimed mined lands meeting specific woody plant densities was thought and remains a major challenge to surface mine operators (Berg,1975). The 1977 Surface Mining Control and Reclamation Act (SMCRA) requires coal mining companies to establish a diverse, effective and permanent vegetative cover on reclaimed lands. The Colorado Surface Coal Mine Reclamation Act, effective (8/30/1980), sets specific criteria for herbaceous cover and production, species diversity and woody plant density to be deemed adequate for final bond release (Mathews,1990).

There is the need for exchange of information on mature shrub transplanting methods, concerns, and successes. Trapper Mine initiated several programs and studies to resolve this woody species density issue.

BACKGROUND

Trapper has experimented with various mature shrub transplanting techniques since 1977. Trials with a tree spade (1977), dryland sodder (1979), scrapers (1982), and 3 models of front-end loaders have been performed (Harding,1990).

Small scale mature shrub transplanting, initiated in 1981, became the basis for the large scale shrub "clump" concept that began in 1982. These shrub clumps are sized at 0.65 ha (1.6 acres), include at least 250 shrub pads, and are spaced approximately 426 m (1400 feet) to maximize wildlife utilization and to minimize the distance to cover within the reclaimed areas dominated by grasses and forbs.

STUDY AREA

The study area is located in the Northwestern corner of Colorado within the federal, state and private coal leases held by Trapper Mining, Inc. at its Trapper Mine. Trapper is a 2.2 million ton/year surface coal operation impacting 3318 ha (8200 acres) through the year 2014. The mine is 10.5 km (6.5 miles) southwest of Craig, Colorado on the northern flanks of the Williams Fork Mountains. The coal beds are part of the Green River-Hams Fork Region within the major land resource area (MLRA) 34 as delineated by the Soil Conservation Service (SCS) and labeled the Central Desertic basins, mountains and plateaus (OSMRE,1988).

The Craig subregion falls within a semi-arid, continental climate regime characterized by dry air, sunny days, clear nights, little precipitation, high evaporation, and substantial diurnal temperature changes. Complex topographic features create considerable variation in site-specific temperature, precipitation and surface winds. Temperatures and local microclimate, to some extent, vary with elevation. Extreme frigid conditions and blizzards can occur but severe weather types such as tornados, floods, and damaging hail are rare. Annual temperatures range from -50 to 38°C (-50 to 100°F) with the mean daily temperatures in January and July -7°C (18.1°F) and 15°C (59.9° F) respectively. The mean annual precipitation is 353 mm (13.9 in). One third falls as snow which is the principle source of the region's stream flow. The mean snow depth is 169 cm (66.5 in).

The soils are derived from sandstone and shale parent materials into deep to moderately deep, well-drained, slightly acid to moderately alkaline sandy loam to loamy soils. Both are formed in alluvium and colluvium deposits. The topography is described as rolling hills with sloping to steep sloping sideslopes. The average slope is 14 percent. The minesite ranges from 1950-2350 meters (6400-7700 feet) in elevation.

The prevalent vegetation is the mountain shrub community comprised of dense stands of mixed shrubs from 0.6 to 2.4 m (2 to 8 feet) in height. This community usually exists as a transition zone between the aspen (Populus tremuloides) and sagebrush (Artemisia tridentata) types. The major shrub species are serviceberry (Amelanchier spp), Gambel oak (Quercus gambelii), chokecherry (Prunus virginiana), and snowberry (Symphoricarpos oreophilus) (Harrington,1954). Scattered stands of antelope bitterbrush (Purshia tridentata), mountain mahogany (Cercocarpus montanus), boxelder (Acer negundo), cottonwood (Populus angustifolia), and juniper (Juniperus spp) exist where moisture, soil depths and aspect are markedly different. This vegetation of many height classes promotes the existence of diverse small mammal and avian populations.

METHODS

Vegetation Clump Transplanting

Mature shrubs are salvaged in advance of topsoil stripping operations and transplanted to preselected and designed sites. Transplanting is done in the fall (September-November) following the first major killing frost or series of frosts. Front-end loaders (FEL's) dig the shrub "pads" out the natural locations transporting them to receiving sites across or around the ends of the active pits.

Each pad must include one of the major transplant species (Prunus or Amelanchier) at least 1.2 m (4 ft) in height and include a minimum of 30 cm (12 in) of topsoil. Each pad should be transported intact to the planting site. Key pads are placed at the design edges and specific points as permanent reference points during the construction process. Lath and other temporary survey markers are readily destroyed by the heavy equipment building the clump.

Mature shrub transplanting at Trapper is based on the vegetative clump method not a single mature tree or shrub transplant. Each transplant "pad" consists of the major overstory species (Prunus or Amelanchier) whether one or more actual shrubs are included, plus all the associated woody subshrubs, perennials, annual herbaceous understory species, seeds and etc. The pad size is determined by the size of the standard or modified bucket of the transporter. Since 1982 Trapper has used various sized models of front-end loaders (wheel loaders) as the only transport system.

Design

Shrub clump placement is done using mine maps (400 scale) developed from annual aerial photographs and field surveys. Clumps are spaced at least 305 m (1000 ft) apart but not greater than 426 m (1400 ft) and above the 2045m (6700 ft) elevation contour. This elevation separates cropland and wildlife reclamation at the mine. Once tentatively designed, the proposed locations are surveyed and examined for micro-topographic features. Slope, aspect, water harvesting and minor surface variations are factors for possible site adjustments. Then 0.8 ha (2 acre) sites are marked for 0.9 m (3 feet) of topsoil replacement and final survey markers installed.

Small scale drawings (50 scale) are made of each site using a 5x10 foot pad size and 250 pads per clump. These drawings incorporate field notes regarding contours, ditches, and physical structures that could affect the clump design (Figure 1). These schematics are also used during clump construction and track the pads placed daily. Each clump is given a four letter alphanumeric code designating the year built, the pit location and a north/south locator since 2 clumps are usually built in the same narrow topsoil replacement area each year. (ie. 86EN is the northern clump in Enfield pit built during 1986).

Transport

Since 1984 Trapper mine has exculsively used front-end loaders as the transport method for mature shrub clump establishment. Additionally after trials with specialized buckets, Trapper has used standard FEL models and buckets which were on site, easily contracted or leased.

Front-end loaders are mobile, versatile, able to handle grades up to 20%, operate effectively on shallow rocky soils (Proctor, 1983), and can handle a wide variety of material (Brown,1977), and are usually in operation at any mine site whether coal,hardrock or a sand/gravel operation. The FEL model used depends on the mine size, topography, and materials to be handled.

They are relatively inexpensive and have short delivery cycles. In load and carry operations FEL's can eliminate the need for other equipment to handle materials. FEL's are agile, perform their own cleanup work and are small enough to selectively load materials (Brown,1977).

Survival Analysis

Survival analysis studies began in 1986. Initially only the dominant overstory species within each pad were examined for live stems. The sampling was to evaluate the effectiveness of the FEL transplant method and to determine the shrub species with the best survival. In 1989 documentation of the entire woody understory component was initiated and recorded in the data.

Following the initial year of establishment, and each summer thereafter, all the shrub clumps and each individual pad are inventoried. All woody species encountered are identified. Overstory regeneration, and sprouting from the pad base are segregated by the three major species (AMLA, PRVI,and QUGA). Snowberry and other woody understory species are denoted. After each pad is examined it is marked with vinyl surveyor flagging and the count/survival status is tabulated.

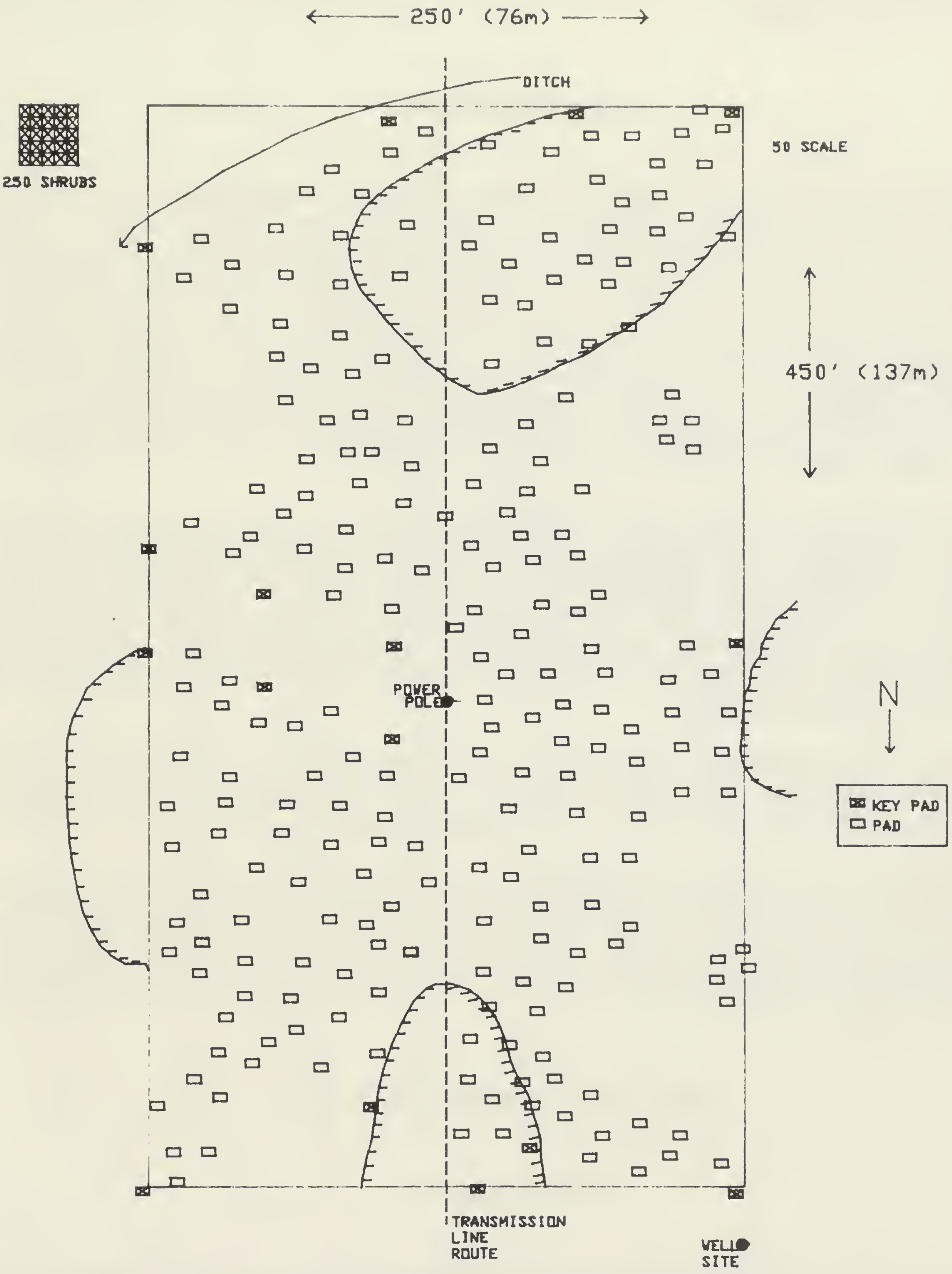

FIGURE 1: TYPICAL TRAPPER MINE SHRUB CLUMP PLANNING DESIGN

Additionally 10 percent of the viable pads within each clump are randomly sampled to determine the total number of stems (by species) represented in each clump. These stem counts are converted to stems/acre estimates for each species, the entire clump, and the woody stem contribution from all shrub clumps to the total woody stem density requirement.

Species Selection

The most important step in revegetating disturbed sites is selecting adapted species. Following proper selection, seeds and/or plants need to be placed on the site with techniques and practices that give every possible opportunity to become established (Stevens,1983). Major factors to consider are native species adapted to the immediate area, soil profiles, rooting characteristics, wildlife and livestock benefit (Dennis,1986), the permitted land use after bond release and the projected land management practices. The final decisions must include the economics involved in both the establishment and ongoing management practices that will insure a self-perpetuating successional series to a stable, viable vegetative community suitable to the post-mine land use and the surrounding environment.

Species selection for mature shrub transplanting is site specific. Trapper selected serviceberry (Amelanchier alnifolia) and chokecherry (Prunus virginiana) as the two species for mature shrub transplanting because 1) they are dominant overstory species, 2) their shallow rooting and sprouting characteristics (Wasser, 1982), 3) the cost and ease of transplanting, 4) their value to wildlife, 5) their tolerance to browsing, or defoliation, and fire, and 6) inherent characteristics for soil stabilization on disturbed areas (Brown,1984).

The other woody subshrubs...big sagebrush, antelope bitterbrush, woods rose, silver and fringed sagebrush, rabbitbrush, currant, juniper and snowberry are additional species provided by the "pad" plus any other ungerminated seed source.

Mature Gambel's oak (Quercus gambelii) weren't utilized in early efforts because of the deep taprooting feature and extreme haulage distances to reclaimed areas in the western half of the mine. Specific transplanting trials began in 1990 using small regrowth (28 pads) and will be attempted from the resprouting oak areas within the prescribed burns of 1989 and 1992.

RESULTS AND DISCUSSION

Effective Moisture

Trapper has found a strong correlation between increased natural precipitation and increased stem densities when evaluating precipitation records for the critical months prior to and following initial shrub clump establishment (Figure 2).

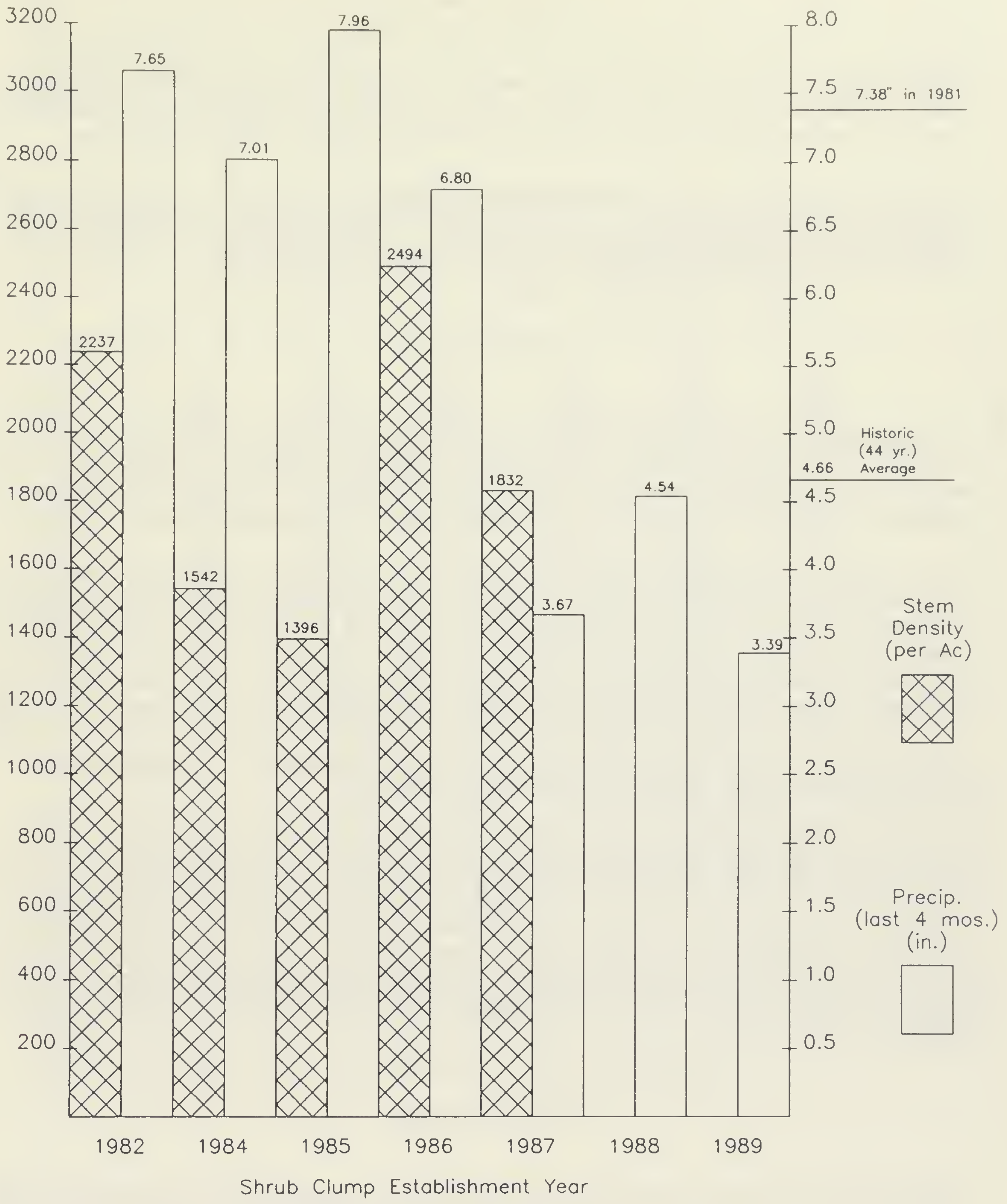

Figure 2. Relationship Between Stem Density and Precipitation Within Mature Shrub Clumps by Establishment Year at Trapper Mine in 1989

Adequate Pad Sizes

Adequate transplant pad size is dependent on the mature plant materials being handled. Different species will require different pad sizes based on the overstory biomass, rooting characteristics, and root biomass to support the overstory size.

Trapper has determined that pad sizes approximating 3 m by 1.5 m (10 x 5 ft) and soil depths ranging from 30-45 cm (12-18 inches) are adequate for acceptable survival levels with chokecherry, serviceberry and snowberry species. All FEL models used, meet or exceed that criteria.

FEL Efficiency

The experimental trials in 1979 with the $10.2m^2$ dryland sodder ($110 ft^2$) indicated the 988 CAT FEL was undersized. The standard 988 FEL buckets in 1980 proved too large for the transplant species as the pads broke apart in transport and additional equipment hours were expended in standing fallen shrubs upright (Hodder, 1976).

Under normal mining operations FEL's aren't economical for hauling distances over 213 m (700 feet). Baker (1981) felt FEL's were inefficient and that transporter systems must be used for distances greater than 300 yds (274 m). Brown (1977) stated transportation distances over one mile have a one-hour cycle time which is inefficient for large scale plantings. However at Trapper they have been consistently used over 762 m (2500 feet) for shrub transplanting while cycle times normally took 25 to 30 minutes. During 1986 a 966 CAT and FR20 FEL took three days to build each clump as the transport averaged over 100 pads/day even with one-way transport distances greater than 762-914 m (2500 and 3000 feet).

In 1990 a rental 950E moved over 500 pads in 29 days. The average was 17 pads/shift and the best was 33 pads/shift (shift = 7.5 hours), or 4.4 pads/hour over round trip distances of 2.3-2.7 km (1.4 miles and 1.7 miles). Once again cycle times were recorded around 25 to 35 minutes. Additionally there were no major wear and tear problems as suggested by Proctor (1983). One problem encountered when moisture conditions were great for pad handling was travel surfaces in remote areas. The FEL had to be chained which slowed production or motor-graders had to "cut" the travel route.

Survival Trends

Overall mature shrub survival precentages (1992) at Trapper Mine range from 36% (84DN) to 99 % (90DN). Although this paper addresses information gathered since 1986 for survival rates, stems/acre for shrub clumps and stems/acre contributions to the total stem density standard, its primary focus is a comparision between the clumps established in 1986 and 1987 with significant differences in the precipitation received prior to, during and upon completion of the projects and the change from the larger $4.5m^2$ 966 model FEL ($50 ft^2$) to the $3.7m^2$ 950B FEL ($40 ft^2$) in 1987.

The total annual survival percentages for the next six years (1987 - 1992) are presented in Table 1. The clumps are identified their alphanumeric code. Two were built in Enfield pit in 1986 and two were built in Derringer pit during 1987. Those built in the southern end of the pits are at the higher elevations (86ES and 87DS) at approximately 2197 m (7200 ft). The northern clumps are usually around 2071 m (6800ft). The precipitation for the last four months of 1986 was 172mm (6.8 in.) vs 93mm (3.67 in.) in 1987 (Figure 2).

Table 1. Total mature shrub pad annual survival percentage for shrub clumps built in 1986 and 1987 at Trapper mine (1987-1992).

Plot ID	1st year	2nd year	3rd year	4th year	5th year	6th year
86EN	98	98	92	71	65	59
86ES	98	98	96	88	95	94
87DN	99	98	82	86	76	--
87DS	97	93	84	87	86	--

The total survivals for all four clumps after two growing seasons are nearly identical ranging between 93 to 99 percent (Table 1). This corresponds with historical survival rates for the first two years. The carbohydrate reserves in both major species are adequate (2-3 years) for leaf regeneration, flowering and usually for fruit production. Transplant shock and overstory vigor declines are normally observed in the third and fourth year after establishment.

There is a marked decline in survival for the 1987 pads during the third season while the larger pads (1986) remain above 90 percent. This decline to an 83% survival average may be correlated to the smaller bucket and drier conditions during establishment. However by the 5th season both survival rates average similarly (80 percent), and correlate with earlier 5th year survival percentages.

Table 2. Live overstory percentages for six growing seasons following mature shrub transplants comparing two sizes of front-end loaders at Trapper mine.

Clump	FEL	Species	Growing seasons 1st	2nd	3rd	4th	5th	6th
86EN	966	AMAL	83	64	34	46	48	10
		PRVI	17	30	24	21	28	16
		QUGA	0	0	0	0	0	0
86ES	966	AMAL	79	73	75	89	78	49
		PRVI	12	20	29	24	35	24
		QUGA	5	1	1	0	0	0
87DN	950	AMAL	48	63	77	71	25	--
		PRVI	18	15	12	14	9	--
		QUGA	0	0	0	0	0	--
87DS	950	AMAL	69	82	81	66	35	--
		PRVI	16	26	25	34	23	--
		QUGA	0	0	0	0	0	--

Live overstory survival percentages for the same six growing seasons show a progressive decline in survival (Table 2). Amelanchier (AMAL) (estab. 1986) is found in 81% of the surviving pads after the first season. Prunus (PRVI) averages 14.5% and Quercus (QUGA) averages 2.5% for the same interval. Within four additional seasons all live QUGA is gone and PRVI is only found within 22% of the pads with live overstory. AMAL vigor recovers briefly for a couple of years but progressively declines through the sixth season. Whereas the AMAL (estab. 1987) averages 58.5 %, PRVI averages 17% and no QUGA are inventoried after one season. Then observations show increased vigor in AMAL the next 3 years to a 79% average with the normal decline to below 35% by the 5th season. PRVI first year survival (15%) is similar to the larger bucket pads and its vigor remains fairly consistent the next four years at approximately 20 percent.

Table 3. Live bottom sprouting percentages for six growing seasons following mature shrub transplants comparing two sizes of front-end loaders at Trapper mine.

Clump	FEL	Species	Growing seasons 1st	2nd	3rd	4th	5th	6th
86EN	966	AMAL	83	69	85	100	69	64
		PRVI	17	31	26	29	37	30
		QUGA	0	0	0	0	1	1
86ES	966	AMAL	86	73	87	100	76	50
		PRVI	8	24	40	40	41	36
		QUGA	6	2	1	1	1	1
87DN	950	AMAL	78	94	82	72	66	--
		PRVI	20	20	17	19	15	--
		QUGA	0	0	0	5	1	--
87DS	950	AMAL	75	95	100	67	62	--
		PRVI	25	45	36	41	32	--
		QUGA	0	1	0	3	1	--

Bottom sprouting inventories for the same four clumps show a different trend (Table 3). Except for clump 86ES, with several original live QUGA overstory and understroy plants there is a gradual increase in basal sprouting by the 4th year. Live AMAL basal sprouting averages 80.5%, in all four clumps the first year, but then declines in the larger pads (2nd year) while increasing in the smaller pads to over 90%. There is almost a 100% basal sprouting surge in all pads during years three and four with the exception of clump 87DN occurring in year two. All clumps decline to approximately 60% thereafter. PRVI averages 22.5% in the smaller pads vs 12.5% in the large pads and that average continues to increase until year five.

Observations show AMAL basal sprouting starting high (75%), then increasing to 100% by the 3rd or 4th year and then declining to 50 - 60% by the 6th year. PRVI sprouting starts about 25% with minor variations until it stabilizes around 30% in year five. QUGA usually are not observed until stems sprout from roots or germinate around year four.

Woody Stem Density Benefit

The Colorado Surface Coal Mining Act(1979) requires mine operators to reestablish woody plants (shrubs) to a density at least 90 percent of the density on an approved reference area. Colorado regulations allow for variances from this reference standard if it can be shown that greater or lesser shrub densities will better serve the postmine land use (Agnew,1992).

Trapper's currently approved total woody stem density is 988 stems/ha (400 stems/acre). Additionally the minimum mature shrub clump density standard has been set at 9,300 stems/ha (3,764 stems/acre).

In 1988 woody stem densities for the 1982-1985 clumps averaged 3,504 stems/ha (1,418 stems/acre) prior to deep topsoil replacement. The woody stem densities for the 1986-1987 clumps averaged 5,812 stems/ha (2,352 stems/acre). The 66% increase (2,803 stems/ha) is attributed to the deeper soils and increased use of supplemental water.

In 1989 the stem densities for the 950B pads (1987) were 1,728 stems/ha less than the larger 966 pads (1986) while a total of 78% (7,255 stems/ha or 2,936 stems/acre) of the minimum shrub clump density standard was obtained.

Costs

The cost for mature shrub transplanting is expensive. All methods of transplanting native vegetation are far more expensive than seeding stated Proctor (1983). Transplanting at Trapper has ranged from $137 per pad (1984) to $40 (1990). The estimated cost for each individual live plant stem within all clumps through 1988 was approximately $ 8.57 (based on an average clump cost of $25,000). Proctor projected costs/0.4 ha (acre) for FEL's at $3.37/plant (1983).

Costs at Trapper prior to 1987 with outside contractors and 966 model FEL's were above $80/pad. Since leasing 950 models and using in-house personnel the costs were reduced over 60 percent ($40 to $60 per pad). Based on 1991's average of 9551 stems/acre (23,600 stems/ha) for the 13 clumps and the 1990 cost per clump ($40/pad), the approximate cost per live stem was $1.47.

Additional cost estimates for particular projects can be derived using the latest edition of the Caterpillar Performance Handbook (1992) sections on "Estimating owning and operating costs." There are even sections entitled "Quick Estimators." Equipment vendors can also provide their latest rental rate information to assist in cost projections.

CONCLUSIONS

Other than a significant cost reduction approximating 50%, comparisions between two years of different moisture regimes and FEL pad sizes indicate the survival trends for AMAL and PRVI are fairly similar. The comparisions between these four clumps seem to average out in the five years available for study.

Stems/clump counts and their projected stems/acre (stems/ha) estimates presently contribute a significant portion (50%) to the total stem density criteria and the 1991 woody stem contributions surpassed the approved clump standard. Further observations will delineate the actual woody component composition as bond release approaches.

Woody species re-establishment on reclaimed mined lands will remain a challenge to mining operations. Operators and the regulatory agencies will need flexiblity in the standards determining an adequate measure for a diverse, effective and permanent vegetative cover and the economics of bond release. No one technique nor practice will accomplish the standard. The selection of adapted species, the methods for transplanting mature shrub or tree species and costs are site specific. Following establishment, a monitoring program must be initiated to determine the treatments and management practices to increase stem densities. There also remains the need for information exchange regarding semi-arid native species transplanting.

SUMMARY

Trapper Mining has constructed 15 mature shrub clumps by the end of 1992. These clumps influence over 279 ha (690 acres) and contribute over 200 woody stems/acre (494 stems/ha) to the total woody stem density requirements for bond release. Overall mature shrub survival ranges from 43 to 100 percent. Since practices of deep topsoil (0.9m) and supplemental watering were instigated survival rates range from 59-100 percent through 6 growing seasons. In 1991 woody stem contributions ranged from 8,540 to 37,621 stems/ha (3,456 to 15,225 stems/acre) while the average of 23,600 stems/ha (9,551 stems/acre), was 250% above the standard required.

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REMEDIATION OF WATER QUALITY PROBLEMS ASSOCIATED WITH DRAINWATER FROM DEPARTMENT OF THE INTERIOR IRRIGATION PROJECTS

R. A. Engberg[1] and A. J. Cappellucci[2]

ABSTRACT

Remedial planning under the Department of the Interior's National Irrigation Water Quality Program (NIWQP) is underway at the Middle Green River area in Utah and the Kendrick Reclamation Project area in Wyoming. Selenium is the principal constituent of concern in each area. In the Middle Green River area median selenium concentrations in all drains to Stewart Lake for data collected between 1986-90, were greater than the Utah criteria of 5 parts per-billion (ppm) for wildlife protection. Selenium concentrations in bird eggs ranged from 2 to 33 parts per-million (ppm) dry weight. A bioaccumulation factor between geometric mean concentrations of selenium in water and in bird eggs was greater than 300 for Stewart Lake. In the Kendrick area, median selenium concentrations for 1986-90 were 54 ppb and 38 ppb for two wetland areas. Geometric mean selenium concentrations all exceeded 30 ppm in bird livers from migratory birds collected at the wetland areas. Deformed migratory bird embryos were found in both the Middle Green and Kendrick areas.

Remedial planning is underway in both areas and will result by 1994 in reports presenting preferred remediation alternatives. Objectives of remediation for both areas include selenium reduction, minimize ecological and human health hazards, and habitat restoration. Remedial options may be categorized as dilution, treatment, source control, disposal, institutional, and no action.

Remedial issues confronting the NIWQP include repayment, and success of remediation. Who plays as well as how to measure remedial success need to be resolved before remedial measures are applied in each area.

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[1] Manager, National Irrigation Water Quality Program, Department of the Interior, 1849 C Street N.W., Washington, D.C. 20240

[2] Irrigation Drainage Program (D-5101), Bureau of Reclamation, Building 67, Denver Federal Center, P.O. Box 25007, Denver, CO 80225

INTRODUCTION

Elevated concentrations of selenium in drainwater reaching Kesterson Reservoir in California and the effects of these concentrations on migratory waterfowl, led to the creation by Secretary of the Interior in 1985 of a Program to determine whether similar problems from selenium of other constituents are associated with other Department of the Interior irrigation projects. Over 10 million acres of irrigated agricultural land receive water from about 600 DOI constructed or managed irrigation projects in the western United States. The National Irrigation Water Quality Program (NIWQP) through 1992 has conducted or has underway reconnaissance investigations at 24 locations in 14 states. Priority locations are areas where National Wildlife Refuges or other wetlands for which DOI has responsibilities under the Migratory Bird Treaty Act or the Endangered Species Act receive drainwater from DOI projects. Because of proximity of small projects with hydrologic similarities, as many as 5 DOI irrigation projects may be included in each investigation.

When reconnaissance investigations determine that problem constituents are present in concentrations of concern, detailed studies are conducted to determine the source, transport mechanisms, and fate of the constituents. This information provides the scientific basis for remedial planning and remediation.

Remedial planning is underway at the Middle Green River area in Utah and at the Kendrick Reclamation Project area in Wyoming. The purpose of this paper is to discuss the magnitude of problems and the remedial planning in these areas and to examine two generic remedial issues.

MIDDLE GREEN RIVER AREA, UTAH

The Middle Green River area is located in northeast Utah. While the Middle Green River Basin drains an area of nearly 45,000 square miles, the part of the basin where remedial planning will be focused is around the Stewart Lake Waterfowl Management Area (WMA) near Jensen, Utah (Figure 1). The Jensen Unit of the Bureau of Reclamation's Central Utah Project provides irrigation water for about 4,000 acres near Stewart Lake WMA. The agricultural land overlies the seleniferous Mancos Shale formation. Soils are derived from this formation and plants known as selenium indicators occur in the area. Five groundwater drains discharge drainwater from about 750 acres of Jensen Unit project land to Stewart Lake.

Problem

Selenium is the principal constituent of concern in the Stewart Lake area. Median selenium concentrations in water samples collected during 198689 from all drains to Stewart Lake were greater the Utah criteria for wildlife protection of 5 parts per billion (ppb). Selenium in water from wells in the vicinity of the drains ranged from less than I to 360 ppb. Only 25 percent of the selenium load entering Stewart Lake was discharged to the Middle Green River; the rest moved into the bottom sediment of Stewart Lake. (Stephens and others, 1992).

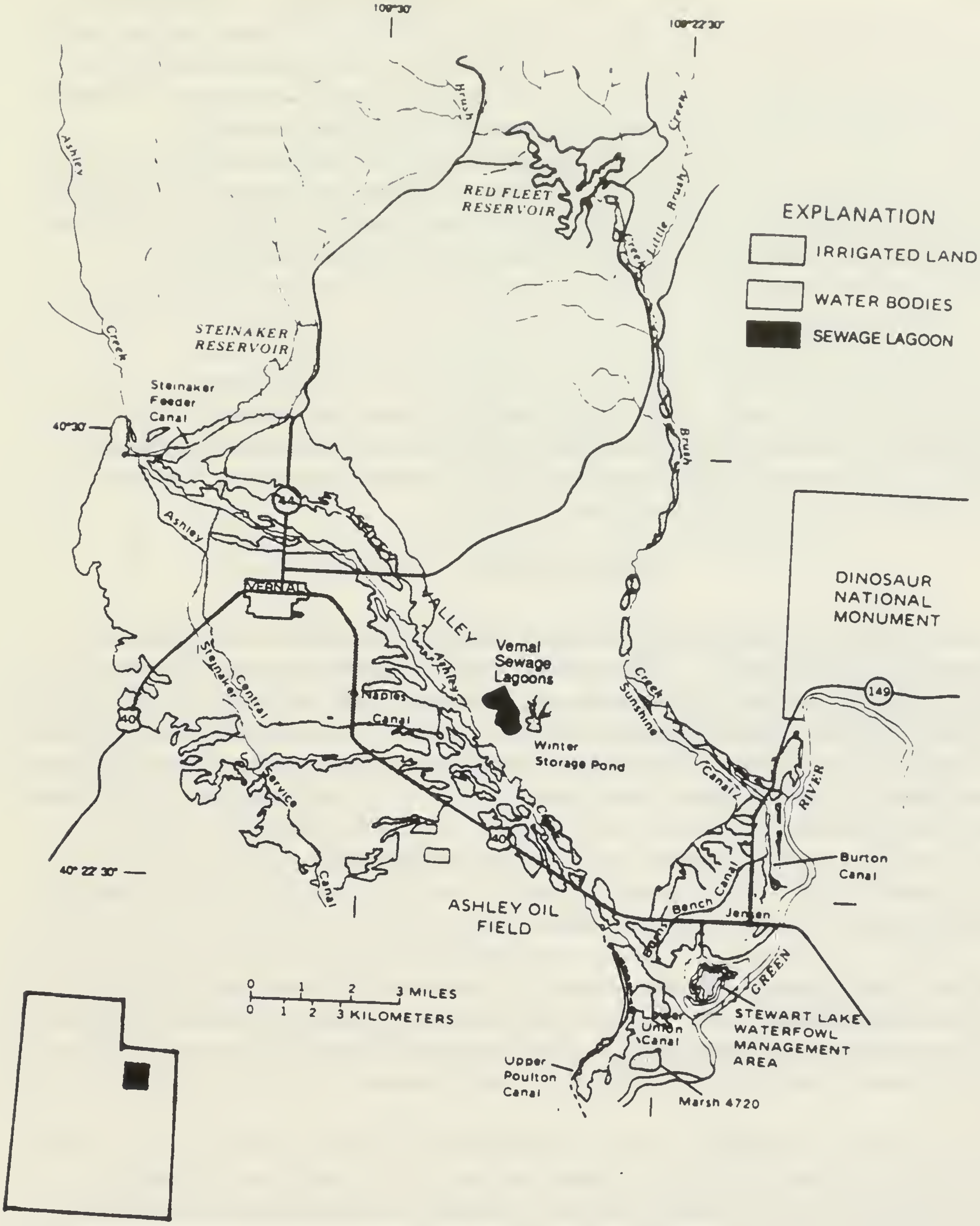


Figure 1. Irrigated land and hydrologic features near Stewart Lake Waterfowl Management area, Utah (from Stephens and others, 1992)

Selenium concentrations in 49 eggs collected in the Stewart Lake area ranged from 2 to 33 parts per-million (ppm) dry weight. (Peltz and Waddell, 1992). Deformed redhead duck and cinnamon teal embryos were observed. Geometric mean selenium concentrations in tissue from 29 birds ranged from 1.9 to 87 ppm dry weight. The geometric mean selenium concentrations in fish from Stewart Lake was 28 ppm dry weight. In an earlier reconnaissance study, the median selenium value in 10 samples of whole body fish was 20 ppm (Stephens, Waddell and Miller, 1988), which was the highest median value of selenium in fish tissue observed in the first 20 reconnaissance investigation by NIWQP. In 1988, the Utah Department of Health issued a health advisory for the Stewart Lake WMA recommending a dietary maximum of no more than 10 ounces of waterfowl or fish per week.

Bioaccumulation of selenium occurs at Stewart Lake. The geometric mean concentration of selenium in 55 samples of drain and lake water was 28 ppb and in whole body fish 24 ppm, a bioaccumulation factor of 850 between water and fish (Stephens and others, 1992). The bioaccumulation factor between the geometric mean concentration of selenium in water and in bird eggs was greater than 300.

Remedial Planning

Planning for remediation began in 1991 for the Middle Green River area and is expected to continue through 1994. The remedial plan is being prepared by an Interagency Study Team led by the Bureau of Reclamation and including participation by the U.S. Geological Survey, the Fish and Wildlife Service, and representatives from State and private entities. The hydrologic and biologic data (Peltz and Waddell, 1992) and the interpretative information from the reconnaissance investigation (Stephens, Waddell and Miller, 1989) and the detailed study (Stephen and others, 1992) provide the scientific basis for remedial planning. In addition, social, economic, legal, and institutional factors unique to the area are being included in the planning process. These factors will be used to assess the impacts of remedial alternatives on the human population of the area.

The plan of study developed by the study team includes the following activities (1) Evaluate the detailed study report, (2) Define (redefine) the problem (3) Identify additional data needs and acquire this information (4) Identify potential remedial alternatives (5) Evaluate alternatives, and (6) Select a preferred remedial plan.

Remedial goals in the Middle Green River Basin study (Interagency Study Team, 1992) include: 1) Reduce selenium in water bottom sediments in the study area which has caused unacceptable hazards to fish and wildlife. 2) Minimize the ecological hazards and satisfy requirements of the Migratory Bird Treaty Act and the Endangered Species Act. 3) Minimize public health risks resulting from consumption of fish and wildlife with elevated selenium levels in the study area.

Public meetings were held several times in 1992 at Jensen, Vernal and Salt Lake City, Utah, to inform the public and solicit remedial options. A notice of intent and scoping was published June 17, 1992, in the Federal Register to comply with regulations of the National Environmental Policy Act (NEPA). Formal NEPA Scoping meetings were scheduled in January 1993. Incorporating the public input, the study team has determined six categories of options (Middle Green River Basin Interagency Study Team, 1992). Several options may be grouped into remedial alternatives. The alternative of choice hopefully will meet the study goals in a flexible and cost-effective fashion.

Categories of options include:

1) No action.
2) Institutional - changes in laws, policies, contracts, or management. These options include fees on applied water, land reclassification, land exchange, land retirement, or grant programs.
3) Disposal - options include removal of contaminated water or sediment. These options include discharge to the Green River, injections well to deep groundwater systems, and removing and disposal of bottom sediments.
4) Treatment - use treatment to reduce contamination. Options include the use of chemical or biological treatment, filtration, evaporation, or ion exchange.
5) Dilution - mixing with higher quality water sources. Options include diversion of more water from the Middle Green River or uncontaminated groundwater to dilute high selenium concentrations.

6) Source Control - improvements in irrigation efficiency and drainage reduction.

A remedial plan with NEPA compliance that includes the selected remedial alternative(s) will be completed in 1994 by the study team. The suggested alternative(s) will be determined by the study team with extensive public involvement and likely will include specific recommendations based on several of the remedial options given above. The remedial plan will address economic institutional and social considerations as well as environmental considerations.

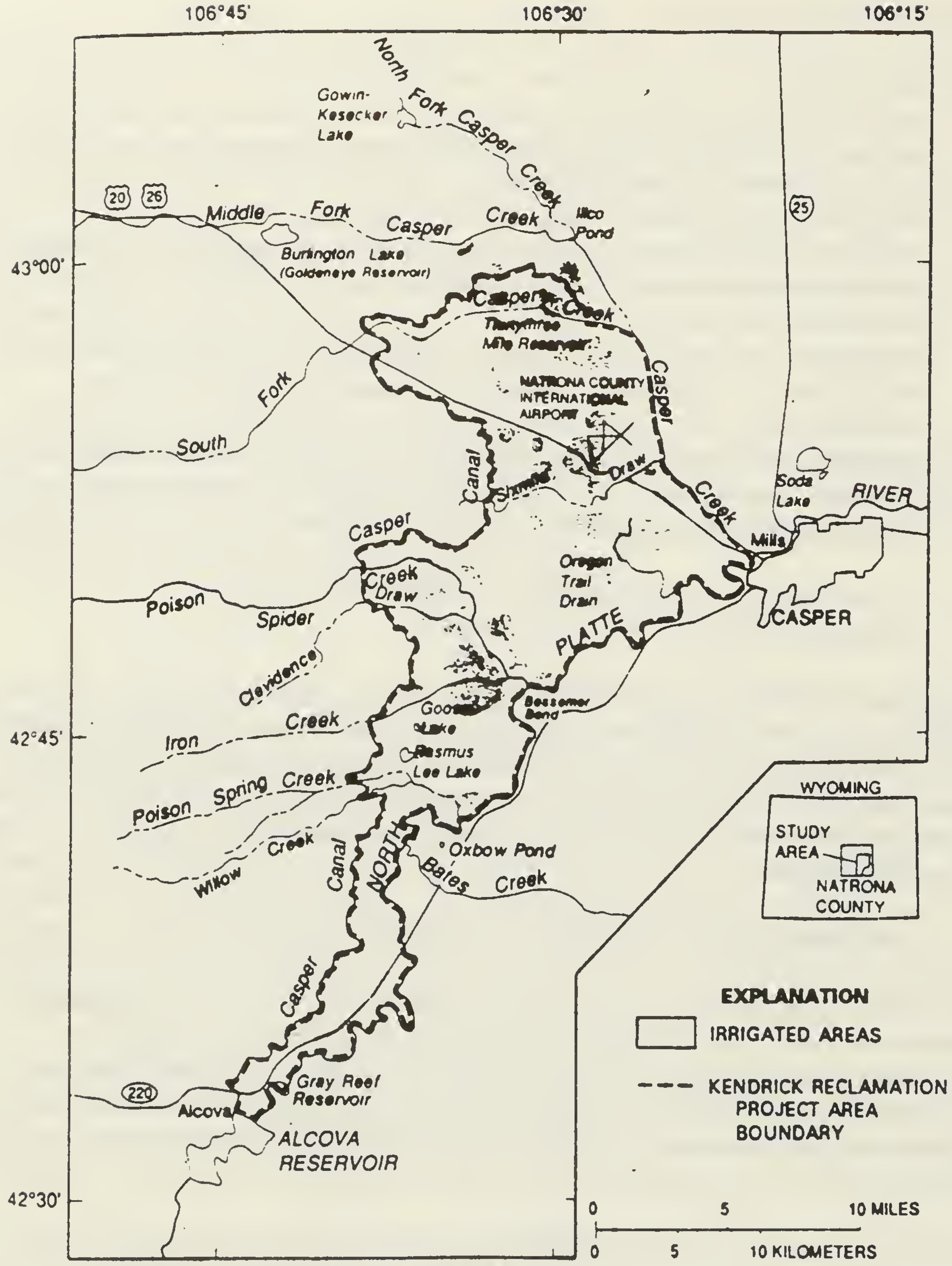
KENDRICK RECLAMATION PROJECT, WYOMING

The Kendrick Reclamation Project study area is located in central Wyoming (Figure 2). Of the 188 square mile area, about 38 square miles receive irrigation water. Water is conveyed to the area by a system of canals and reservoirs from Alcova Reservoir on the North Platte River. Drainwater returns to North Platte River via several creeks. Many of the soils in the area are derived from the seleniferous Cody Shale. Plants known as selenium indicators occur in the area.

Problem

A reconnaissance investigation conducted in 1986-87 (Peterson and others, 1988) and a detailed study conducted in 1988-90 (See and others, 1992) identified selenium as the principal constituent of concern in the Kendrick area. Median selenium concentrations in Poison Spring Creek which receives drainwater and conveys it to the North Platte River, were greater than I ppb upstream of the irrigated area and 92 ppb downstream of the irrigated area. This indicates that the applied irrigation water mobilized selenium from the soil and transported it to the drain. Similar relationships were observed in other drains. The principal water bodies within the Kendrick project area are Goose Lake and Rasmus Lee Lake. The median selenium concentrations in Goose Lake, (15 samples) and Rasmus Lee Lake (27 samples) were 54 ppb and 38 ppb respectively. During periods of low rainfall, these lakes are closed drainages.

Geometric mean selenium concentrations in all migratory birds livers collected at Goose and Rasmus Lee Lakes exceeded 30 ppm dry weight. Deformed migratory bird embryos were found in the area and nesting success was considered to be low.



Base modified from U.S. Geological Survey 1:100,000 digital line graph map series: Casper, 1979, and Midwest, 1981
Universal Transverse Mercator projection

Figure 2. Location of the study area and the Kendrick Reclamation Project area, (From See and others, 1992)

Bioaccumulation occurs at Goose and Rasmus Lee Lakes. Geometric mean selenium concentrations in biota samples ranged from 900 to 3000 times greater than median selenium concentrations in water (See and others, 1992).

Remedial Planning

Remedial planning for the Kendrick area began in 1991 and like the Middle Green area will continue through 1994. An Interagency Study team comprised similarly to the Middle Green study team receives input and assistance from the Wyoming Governor's Selenium Task Force. The hydrologic and biologic data (See, Peterson and Ramirez, 1992), and the interpretations from the reconnaissance investigation (Peterson and others, 1988) and the detailed study (See and others, 1992) provide the scientific basis for remedial planning. Social, economic, legal, and institutional factors from the area will be used to assess human impacts of remediation.

General objectives of remedial planning (Kendrick Interagency Study. Team, 1992) include:

1) Protect and restore migratory bird habitat
2) Identify and minimize potential public health risks
3) Address existing and future uses of water
4) Address social and economic implications

Public meetings were held in Casper, Wyoming, in June and October 1992, to present information and to receive input from the public on remedial options from similar categories of options to those generated by the Middle Green study teams and public. The Kendrick study team and the public will determine preferred remedial alternatives and the team will include these alternatives in a remedial plan to be completed in 1994 along with the recommended alternatives. The evaluation process leading to the remedial plan will address economic considerations as well as environmental considerations, and will produce a plan designed to meet the objectives while maintaining flexibility, cost effectiveness, and satisfying the public concerns.

REMEDIAL ISSUES

Two issues that arose during the remedial planning for the Middle Green and Kendrick areas have not been resolved. The two issues to be discussed here can be phased as questions:

1) Who pays for remediation?
2) What measurements can be used to determine the success of remediation?

Repayment

The issue of repayment may well determine whether remediation takes place. The repayment issue has not yet been decided with regard to who pays for the planning and cleanups of Kesterson Reservoir and the research and planning of the San Joaquin Valley Drainage Program. In 1991, the Secretary of the Interior was directed by Congress to review repayment policies for these areas and to determine a repayment formula. This has not been completed. Until liability for cleanup costs has been established, Congress directed that DOI will not recover planning or cleanup costs from irrigation districts associated with these areas. Resolution of this issue is necessary before the repayment issue can be addressed for the Middle Green and Kendrick or other NIWQP study areas.

The reason repayment is an issue relates to the way in which DOI is involved in the construction of irrigation projects. Under Reclamation law, an irrigation district initiates a project and requests that DOI construct the project. If feasibility and cost-benefit analyses look favorable, DOI will sign agreements with the irrigation district to construct the project. The district agrees to repay DOI for construction costs with interest over a future period. The district also agree to repay DOI operation and maintenance costs on an annual basis when DOI operates the project. If is not clear who has liability for cleanup costs in situations where environmental damage occurs as a result of drainwater or other project related activities, i.e., is remediation a legitimate operation and maintenance cost. The question is therefore, are the cleanup costs wholly a DOI responsibility, wholly an irrigation district responsibility, or in some percentage the responsibility of both?

Success Of Remediation

How to determine the success of remediation for each area may be extremely difficult. Remedial measures impact the hydrology of the area but a determination of remedial success also involves biological measurements. A few of the possible questions upon which measurement of remedial success may depend are: Have the selenium concentrations decreased in migratory migratory waterfowl? Have identified impacts been reduced to levels of no impact and what are these levels? Has nesting been more successful? Have overall populations of birds stabilized or improved?

Because many proposed remedial options will impact or alter the hydrologic system, it has been suggested that a measurable parameter of remedial success is the concentration of the constituent of concern in water in the project area. Remediation based on concentration of a constituent (i.e. selenium) could be approached in three ways: (1) an attempt to remediate to an existing environmental standard or criteria, (2) to remediate to a concentration at which no impacts occur to the biological species in question, or (3) to remediate to a negotiated concentration of the constituent.

Remediation to a single standard or criteria may be difficult in that several choices of standards or criteria may be available. For example, criteria for individual states may differ from each other and from Federal standards or criteria. Second, impacts vary from species to species based on characteristics and diet. Third, few of the available standards or criteria are based on actual field conditions. Rather, the standards or criteria often are based on single species data derived from laboratory experiments that do not incorporate field conditions and do not assess effects of other constituents or environmental factors. Even if standards or criteria based on field conditions were generated for one area, they might not be appropriate for other areas where field conditions are different. Finally, in some areas naturally occurring concentrations of the constituent in question may exceed standards or criteria.

Remediating to a level of no impact on biological species is an alternative to remediating to standards or criteria. Because this would be based on impacts in individual areas, the levels of impact of constituents likely are variable from area to area. This approach also depends on environmental conditions such as climate because climatic fluctuations may affect the availability of the constituent of concern to the hydrologic system. A sophisticated monitoring program would be necessary to determine the point at which impacts cease to occur. The monitoring program would have to continue indefinitely to verify lack of impact. The cost of long term monitoring combined with remedial measures to achieve lack of impacts may be prohibitively expensive.

An alternative to remediation to criteria or lack of impact is the establishment of negotiated target remedial concentrations of individual constituents for individual areas. These targets would be determined through a consensus of responsible entities and organizations in the area and would be based on conditions in the individual areas. Cost effectiveness may play a large role in determining these targets since cleanup costs generally are not straight line but often grow exponentially related to the amount or degree of cleanup. These remedial goals may be a balance between the costs and the benefits of the remediation. Determining targets by consensus may require relaxation of some criteria or waivers of existing laws to achieve the remedial goals.

No decision has been made by NIWQP on the remedial approach that will be favored by the Program. Indeed, different approaches may be appropriate for each area ultimately remediated by the Program . Realistically, remediation will depend on the legal responsibilities, on the availability of funding, and on the will of the people in the impacted areas.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

COMPUTER SIMULATIONS OF THE INFLUENCE OF SUSPENDED IRON-RICH PARTICULATES ON TRACE-METAL REMOVAL FROM MINE-DRAINAGE WATERS

Kathleen S. Smith[1], Walter H. Ficklin[1], Geoffrey S. Plumlee[1], and Allen L. Meier[1]

ABSTRACT

Many existing remediation strategies for the treatment of mine drainage involve oxidation and (or) neutralization of mine-drainage waters. Iron-rich particulates are commonly formed as a result of the oxidation and (or) neutralization of these waters. These iron-rich particulates have the capacity to sorb trace metals contained in the mine drainage. The sorption reactions are very sensitive to the pH of the mine-drainage water and also to the ratio of the concentration of suspended iron-rich particulates relative to the concentration of dissolved trace metals. A method is presented here that applies computer simulations to the prediction of trace-metal sorption by suspended iron-rich particulates, and case studies are modeled. This method can be used to predict the removal efficiency for trace elements in oxidizing-neutralizing natural and mine-drainage treatment systems. The importance of pH and particulate/trace-metal ratio on the amount of trace metals sorbed onto particulates is demonstrated in the case studies. Generally, sorption of trace-metal cations increases with higher pH; in some instances, competition effects between sorption and dissolved species decrease the amount of trace metals sorbed at pH values greater than about 7.5. In the presence of a moderate amount of suspended iron-rich particulates, arsenic, lead, and copper tend to be predominantly sorbed at neutral pH, whereas zinc, cadmium, and nickel tend to remain dissolved. In the presence of an abundant excess of suspended iron-rich particulates, however, most trace-metal cations tend to be predominantly sorbed at neutral pH.

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[1] U.S. Geological Survey, Denver Federal Center, MS 973, Denver, CO 80225-0046

INTRODUCTION

Drainage from metal-producing mining areas is commonly characterized by acidic conditions and elevated concentrations of dissolved sulfate, iron, aluminum, manganese, arsenic, zinc, copper, lead, cadmium, and other trace metals. The acidic conditions result from oxidation of iron-rich sulfide minerals in tailings, mine dumps, and mine workings. The elevated metal concentrations result from the oxidation of sulfide minerals (producing metals such as copper, lead, and zinc) and acid dissolution of rocks (producing metals such as aluminum and manganese), commonly referred to as acid rock drainage. A number of geological and geochemical factors control the chemical composition of mine-drainage waters (Wildeman et al., 1974; Ficklin et al., 1992; Plumlee et al., 1992, 1993; Smith et al., 1992). Understanding these controls is important because many of the trace metals in mine-drainage waters pose health risks and contaminate receiving waters. In order to prevent this contamination, mine-drainage waters must be treated to remove, immobilize, and (or) concentrate trace metals.

Chemical treatment strategies for remediation of mine drainage commonly involve increasing the pH (neutralization) of the drainage water. To raise the pH, materials such as sodium hydroxide (caustic soda), calcium hydroxide (lime), ammonia, calcium phosphate (rock phosphate), calcium carbonate (limestone, calcite) (Lapakko and Antonson, 1991), or calcium-magnesium carbonate (dolomite) are added to the mine drainage. It is commonly necessary to raise the pH to a value of 9 or 10 in order to effectively remove most trace metals. At these high pH values, most trace metals precipitate as solid hydroxides (Clifford et al., 1986). An iron-rich sludge is generally also formed by these neutralization-precipitation treatment processes (Kuyucak et al., 1991). Other treatment techniques chemically, physically, or biologically remove metals from mine drainage waters (Kondos et al., 1991).

Much interest has been expressed in passive methods to treat mine drainage. These methods are less costly and involve much less maintenance than do more active chemical treatment strategies. Commonly, passive methods involve restricting the flow of mine-drainage water to allow oxidation and precipitation reactions to take place. This can be accomplished by construction of (1) a series of settling ponds or (2) short residence time, surface-flow aerobic wetlands (Brodie et al., 1988). Mechanisms that control trace-metal retention in these systems are generally poorly understood, and these remediation techniques are typically less efficient in the removal of trace metals than are active chemical treatment strategies (Kleinmann et al., 1991). Many of these remediation techniques were originally developed to treat coal-mine drainage (Wieder, 1989). In coal-mine drainage, the metals of concern are usually limited to iron, manganese, and aluminum. Few coal-mine drainages contain the elevated trace-metal concentrations commonly found in metal-mine drainages. Consequently, if these techniques are to be applied to the remediation of metal-mine drainages, it is critical to understand, and to be able to predict, trace-metal behavior in these systems.

Sorption reactions are known to control dissolved metal concentrations in many natural systems. Sorption is the process by which a dissolved constituent is lost from the solution phase and then associated with a contiguous solid phase. Hence, sorption is a mechanism by which trace metals can be immobilized as long as the solid phase is immobile. Detailed discussions of sorption processes are given by Dzombak and Morel (1987), Davis and Kent (1990), Stumm (1992), and Smith (1993).

Most mine-drainage waters (both coal- and metal-mine waters) contain large amounts of iron. As the pH of the water is raised and (or) oxygen is added, dissolved iron precipitates as hydrous-oxide minerals. Hydrous iron oxides readily sorb trace metals in well-defined laboratory systems (Swallow et al., 1980; Benjamin and Leckie, 1981). Computer models have been developed to simulate these sorption reactions

(Davis and Leckie, 1978; Dzombak and Morel, 1990). It has been demonstrated that these models can adequately predict trace-metal partitioning between water and sediment in iron-rich metal-mine drainage systems (Smith et al., 1989, 1991; Davis et al., 1991; Smith, 1991; Smith and Macalady, 1991). The sorptive properties of these hydrous iron oxides depend on the pH of the surrounding water, the chemical composition of the water, and the ratio of the amount of dissolved trace metals to the amount of hydrous iron oxides (Dzombak and Morel, 1987; Davis and Kent, 1990; Stumm, 1992; Smith, 1993).

Smith et al. (1992) observed that in several diverse systems metal-mine drainage waters appear to be equilibrated with iron-rich suspended particulates rather than with iron-rich bed sediment. This is likely to also be the case in restricted-flow systems such as settling ponds. Reasons that dissolved constituents are not in equilibrium with the bed sediments may include (1) hydrologic controls that prevent equilibration of dissolved constituents with the bed sediment, (2) the presence of a stagnant liquid film at the water/bed-sediment interface that interferes with diffusion of dissolved constituents, (3) the physical proximity of suspended sediment to dissolved constituents, and (4) differences in sorption site-binding energies between the suspended particulates and the bed sediment. It appears that suspended particulates are an important solid phase for sorption of trace metals in many mine-drainage systems. For treatment purposes, contact between mine-drainage waters and suspended iron-rich particulates should be maximized to enhance sorption of trace metals. These particulates must then be removed from the mine-drainage waters to immobilize the sorbed trace metals.

We describe a computer-simulation methodology that can be used to predict sorption reactions between suspended iron-rich particulates and acidic mine-drainage waters. Simulations have been applied to case studies to predict the mobility of arsenic, lead, copper, zinc, cadmium, and nickel under conditions of changing pH and chemical composition.

COMPUTER-SIMULATION METHOD

Computer simulations were conducted with MINTEQA2 (Allison et al., 1991). This program is available through the U.S. Environmental Protection Agency Center for Exposure Assessment Modeling, Athens, Georgia. It is an equilibrium chemical-speciation model, coupled with the generalized two-layer sorption model (GTLM) (Dzombak and Morel, 1990). These programs, combined, simultaneously compute sorption reactions and solution equilibria.

The GTLM is a type of surface-complexation model that can predict sorption behavior over a wide range of pH and water composition. Surface-complexation models assume that sorption reactions take place at coordination sites on the solid surface. The binding of a metal ion by these sites is strongly pH dependent, and the pH interval in which a metal ion binds to these sites varies for different metals and for different surfaces. The concentration of these surface sites can be estimated for a given solid, and equations can be written and complexation constants determined to describe various coordination reactions between the surface sites and the metals in question. Dzombak and Morel (1990) have assembled a database of surface-complexation constants for metal sorption onto hydrous ferric oxide (HFO). HFO is a compound to which physical, chemical, and surface properties have been ascribed based on laboratory and literature data (Dzombak and Morel, 1990). The database of HFO surface-complexation constants and accompanying properties for HFO were used in our computer simulations.

For each computer simulation, complete analytical information on water composition was input to MINTEQA2. The water had been filtered to 0.1 μm to remove suspended particulates. For a given water composition in the simulations, the pH was fixed at desired values (incremental values to as much as pH 8) with the water in equilibrium with atmospheric carbon dioxide. Iron and aluminum were allowed to precipitate during the simulations in order to control their solubilities. The mechanism for removal of trace metals in the simulations was sorption onto HFO.

HFO was the sole sorbent material in the computer simulations. To calculate the amount of HFO present in each simulation, the dissolved iron contained in each water was assumed to precipitate. The amount of HFO sorbent material was calculated from this original dissolved iron content. This assumption was made in order to simulate the formation of suspended iron-rich particulates during progressive neutralization of mine-drainage water. As mentioned earlier, neutralization is one of the common strategies in the treatment of mine drainage, and suspended particulates are an important solid phase for sorption of trace metals in many mine-drainage systems.

It should be emphasized that all of the model input was obtained either from chemical analysis or from information given in Dzombak and Morel (1990). None of the model parameters were modified for these simulations, and no fitting parameters were used.

CASE STUDIES

Three different case studies were predictively modeled to demonstrate the importance of suspended iron-rich particulates in neutralization remediation practices. The model-derived sorption prediction results are shown in Figures 1 - 3 as a function of pH. The computer model increased the pH stepwise from ambient to pH 8. Each case presented here represents a different type of mine-drainage composition based on the classification scheme of Ficklin et al. (1992). Details about the water chemistry and geology at each site are given in Plumlee et al. (1993, this volume).

Case 1: Very Low pH and High Trace-Metal Concentrations with Moderate to High Dissolved Iron Concentration

The example for this case is water from the Argo Tunnel in Idaho Springs, Colorado. The Argo Tunnel drains mine workings that exploited pyrite-rich gold-quartz veins in the Central City mining district. This pH 2.9 water contains 120 mg/L dissolved iron. Figure 1 illustrates predictive modeling results if this 120 mg/L dissolved iron precipitates as HFO and is available to sorb dissolved trace metals over the pH range 3 to 8 in equilibrium with atmospheric carbon dioxide. Sorption of cadmium over the pH range 3 to 8 was <4 percent, and sorption of nickel was <6 percent; therefore, these are not shown in Figure 1.

In this case, there is a relatively low concentration of surface metal-coordination sites on HFO compared to the high concentrations of dissolved trace metals, especially zinc and copper. Hence, the number of surface sites available for sorption of trace metals is limited. Figure 1 shows that most of the arsenic, lead, and copper and about 20 percent of the zinc are predicted to be scavenged by HFO at pH 8. The amount of sorbed copper and lead decreases between pH 7 and pH 8, however, due to competition effects from aqueous species (mostly hydroxide and carbonate species). Based on these model predictions, dissolved zinc, cadmium, and nickel would be expected to be mobile at this site over the pH range of 3 to 8.

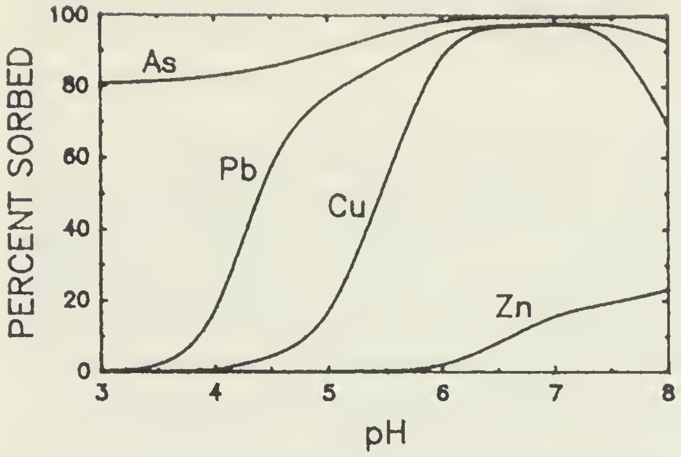

Figure 1. Computer simulation of trace-metal sorption onto hydrous ferric oxide for water composition from the Argo Tunnel, Idaho Springs, Colorado.

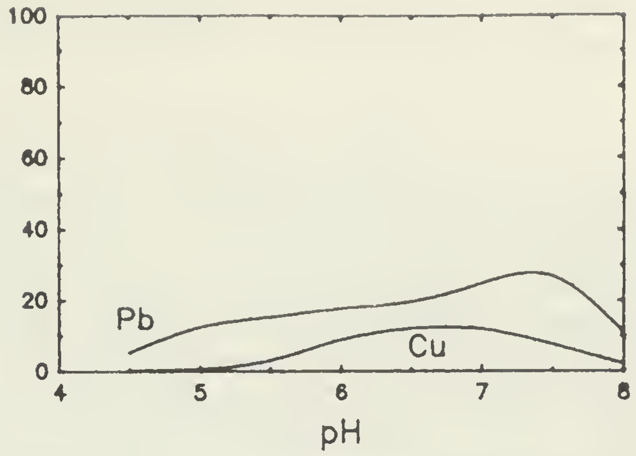

Figure 2. Computer simulation of trace-metal sorption onto hydrous ferric oxide for water composition from the Yak Tunnel near Leadville, Colorado.

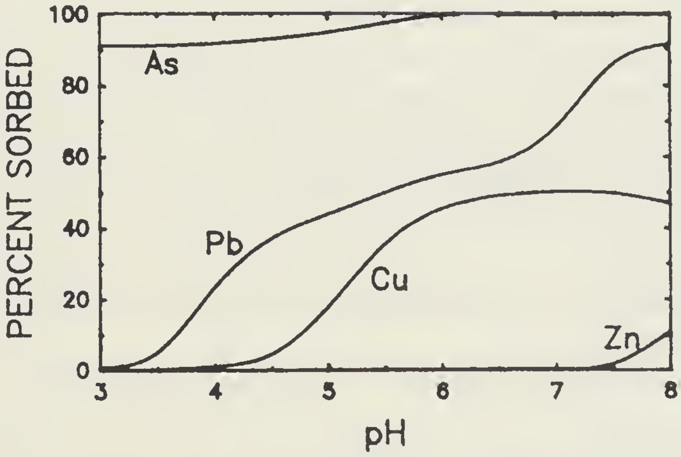

Figure 3. Computer simulation of trace-metal sorption onto hydrous ferric oxide for water composition from the Reynolds Tunnel, Summitville, Colorado.

Case 2: Low pH and High Trace-Metal Concentrations with Moderate to Low Dissolved Iron Concentration

The example for this case is water from the Yak Tunnel near Leadville, Colorado. The Yak Tunnel drains workings in the Leadville mining district that were developed primarily within pyrite-rich polymetallic veins and disseminated ores hosted by igneous porphyry sills. A lesser component of the Yak drainage originates from workings in high-sulfide carbonate-replacement orebodies. The pH of this water is 4.4, and it contains 2.4 mg/L dissolved iron. Figure 2 illustrates predictive modeling results if this 2.4 mg/L dissolved iron precipitates as HFO and is available to sorb dissolved trace metals over the pH range 4.5 to 8 in equilibrium with atmospheric carbon dioxide. Arsenic was not detected in this water. Sorption of zinc, cadmium, and nickel over the pH range 4.5 to 8 was <1 percent (curves not shown in Figure 2).

In this case there is very little HFO available for trace-metal sorption relative to the high concentrations of dissolved trace metals, especially zinc. Hence, the number of surface-coordination sites available for sorption is severely limited. Figure 2 shows this severe surface-site limitation in that no trace metals are significantly retained by HFO at pH 8. Only small amounts of lead and copper are predicted to be sorbed by HFO over the pH range 4.5 to 8. Consequently, dissolved lead, copper, zinc, cadmium, and nickel would be expected to be mobile at this site over the pH range 4.5 to 8.

Case 3: Very Low pH and Extreme Trace-Metal Concentrations with High Dissolved Iron Concentration

The example for this case is water from the Reynolds Tunnel at Summitville, Colorado. The Reynolds Tunnel drains acid-sulfate epithermal ores of the Summitville mining district. The Summitville ores are rich in pyrite and contain abundant copper sulfide minerals and copper-arsenic sulfide minerals. This pH 2.9 water contains 310 mg/L dissolved iron. Figure 3 illustrates predictive modeling results if this 310 mg/L dissolved iron precipitates as HFO and is available to sorb dissolved trace metals over the pH range 3 to 8 in equilibrium with atmospheric carbon dioxide. Sorption of nickel over the pH range 3 to 8 was <1 percent, and sorption of cadmium was <2 percent (curves not shown in Figure 3).

Even though a large amount of particulate iron is available to sorb trace metals, there are again a limited number of surface-coordination sites. This is due to the saturation of surface-coordination sites on HFO as a result of the extremely high concentrations of dissolved trace metals, particularly copper and zinc. Figure 3 shows that most of the arsenic and lead are predicted to be scavenged by HFO at pH 8. Less than 50 percent of the copper, very little zinc, and virtually no cadmium and nickel are predicted to be retained by HFO. Hence, dissolved copper, zinc, cadmium, and nickel would be expected to be mobile at this site over the pH range 3 to 8.

CONCLUDING REMARKS

The simulations presented herein illustrate the potential predictive power of computer modeling. This approach can provide guidance in mine-drainage remediation and planning efforts. Application of these predictive models to natural systems is in its infancy, however, and requires further field validation and evaluation to define the limitations of this approach. Also, competition by trace elements not considered in this work may affect (and likely reduce) the sorption of those trace metals reported here. Sorptive behavior of a variety of trace elements in mine drainage should be further studied and the resulting information incorporated into computer models.

Sorption of trace metals by suspended iron-rich particulates is an important aspect of mine-drainage remediation techniques that involve neutralization and (or) oxidation of iron-rich waters. Mine-drainage pH is very important in determining these sorption reactions, as is the concentration of suspended iron-rich particulates relative to the concentration of dissolved trace metals. In the presence of an abundant excess of suspended iron-rich particulates, most trace-metal cations tend to be sorbed at neutral pH; however, when these particulates are not in abundant excess, differences in trace-metal sorption as a function of pH are apparent. These differences in the pH regime for sorption of a particular trace-metal ion can be used to estimate the pH necessary for removal of that trace metal and, conversely, to predict the degree of mobility of that trace metal. These differences could also be used to design a treatment system for the selective recovery of specific trace metals from water.

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COVER/SOIL EFFECTS ON RECLAIMED MINELAND EROSION

S. A. Schroeder[1]

ABSTRACT

Reclamation laws require soil erosion rates after reclamation to be minimized to prevent off-site contamination of undisturbed areas and on-site degradation of the reclaimed area. The objective of this research was to develop a relationship between basal cover and soil loss on reclaimed minelands. Artificial rainfall was applied at a targeted rate of 57 mm h^{-1} over a succession of runs and soil loss was measured. Cover ranged from 0 to 100% on various reclaimed areas of differing soils and slope gradients. Soil losses were adjusted for factors in the Universal Soil Loss Equation including soil erodibility from nomograph estimates. Linear relationships were found between cover and adjusted soil loss values for each application run and for run data weighted in a manner similar to that used for soil erodibility. These relationships should be useful for determining the amount of cover needed to minimize erosion rates to comply with regulatory requirements.

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[1] Land Reclamation Research Center, North Dakota State University, Box 459, Mandan, North Dakota 58554.

INTRODUCTION

Studies using both natural and simulated rainfall to determine erosion from agricultural soils have been extensive within the last 50 years. Until the last decade, little research on reclaimed minelands was undertaken to determine mining and reclamation effects on erodibility of reclaimed soils. Much of the data from undisturbed soils has been incorporated into models such as the Universal Soil Loss Equation (USLE) to predict average annual soil losses (Wischmeier and Smith, 1978). More recently collected data has been used in the revised USLE (RUSLE) and Water Erosion Prediction Project (WEPP).

Research on reclaimed minelands has been used, for example, to modify variables in the USLE such as the slope factor (Hahn et al., 1985). Other soil loss studies on reclaimed minelands have investigated such factors as grazing intensity effects (Hofmann et al., 1983) and comparative losses between reclaimed and undisturbed grasslands (Schroeder, 1989).

Both federal and state laws require that erosion rates on reclaimed minelands be similar to undisturbed sites. This is necessary to prevent contamination of undisturbed soils by eroded sediment from reclaimed areas and to minimize degradation of the reclaimed site itself. The objective of this research was to develop a relationship between soil losses from reclaimed minelands and basal cover to help predict the amount of cover necessary to control erosion to acceptable limits.

METHODS AND MATERIALS

Reclaimed sites at the BNI Coal LTD mine near Center, ND, and the Consolidation Coal Company mine near Stanton, ND, were chosen based upon differences in vegetative cover, slope gradient and soil texture. All revegetated plots had been established naturally without irrigation. The wide range of cover resulted in differences among plots for age of vegetation (from seeding date) and soil replacement depths (from differences in reclamation requirements). Generally speaking, as the age of the reclaimed sites increased so did cover. It was not possible to distinguish effects of age/replacement depths in this study. Bare plots used were either newly respread and tilled with chisel/disk/harrow or vegetated sites that were rototilled and hand raked to remove cover.

Two replicated plots, 1.8 by 4.9 m, were installed for each slope gradient within the various reclaimed sites. Plots were enclosed with steel borders to a depth of approximately 50 mm. Percent basal cover (live + litter + rock) within the plots was determined prior to the simulated rainfall applications with a vertical point frame (Hofmann et al., 1983). The dominant species on the sites included western wheatgrass (*Agropyron* *smithii* Rydb.), intermediate wheatgrass (*A. intermedium* Beauv), Kentucky bluegrass (*Poa pratensis* L.). and alfalfa (*Medicago sativa* L.). Many others were present in smaller amounts. The vegetation was 0.3 m in height or less on all sites except 6 plots on one site where it approached 0.5 m in height.

An overhead-rail rainfall simulator as described by Schroeder (1989) was used to apply the simulated rainfall at a target intensity of 57 mm h^{-1} (approximate kinetic energy of 0.21 MJ mm^{-1} ha^{-1}). A tarp enclosed three sides of the simulator to minimize wind effects. Some variation in amounts of water applied still occurred due to wind and mechanical problems. The sequence of simulated rainfall application was an initial (dry) run of 60 min at antecedent soil moisture followed by two 30-min runs (wet and very wet). Each run was separated by a 30-min time interval.

Soil cores were removed prior to the dry run at 0-50, 50-100, 100-150, and 150-300 mm depth segments to determine bulk density, organic matter, particle size determination, and antecedent moisture. Soil erodibility K values were estimated for each replicated plot using the soil characteristic values from the 0-50 mm soil core samples and the USLE nomograph (Wischmeier and Smith, 1978).

Runoff from the simulated rainfall applications was measured using a precalibrated 0.18 m HS flume with an attached stage recorder. Aliquot samples were collected for each 10 min interval during each run for sediment concentration, adjusted for runoff volume, and summed for soil loss calculations. An adjusted soil loss factor (ASL, unitless) for each run was calculated from the measured plot data using the following equation:

$$ASL = \frac{Soil\ Loss}{EI * LS * K}$$

where EI is the kinetic energy of the actual rainfall amount applied and LS and K are the USLE terms for slope length/gradient and soil erodibility, respectively. Additionally, a weighted adjusted soil loss factor (WSL, unitless) for each plot was calculated in a manner similar to K (Wischmeier et al., 1971) by the following equation:

$$WSL = \frac{13\ A_d + 4\ A_w + 3\ (A_w + A_{vw})}{13\ EI_d + 4\ EI_w + 3\ (EI_w + EI_{vw})}$$

where A is total soil loss divided by the USLE terms for K and LS. The subscripts d, w and vw refer to the dry, wet, and very wet rainfall simulation runs, respectively.

A dimensionless cover factor for each plot was calculated as follows:

$$Cover\ Factor = \frac{100 - \%\ Cover}{100}$$

Boundary values of 1 for bare plots and 0.011 for plots with 100% cover were imposed upon the values in a manner similar to those used in the USLE (Wischmeier and Smith, 1978).

Values for ASL and WSL (dependent variable) were correlated to the cover factor (independent variable) using PROC CORR (SAS Inst., 1990). Three data points were discarded because of equipment malfunctions that affected the collected data leaving 43 data points in the analyses.

RESULTS AND DISCUSSION

The widest ranges in particle-size distribution of the replaced topsoil among the reclaimed sites were for sand and silt (Table 1). Bulk density tended to be higher on the newly reclaimed sites. The majority of bulk density values ranged from 1.1 to 1.4 Mg m^{-3}. Cover tended to increase with age.

The estimated nomograph K values (Table 1) also reflect the range in particle-size distribution and organic matter among the sites. Most plots had structure classifications of very fine or fine granular. Permeability classes were estimated using the infiltration rate calculated from the runoff rates near the end of the wet run rainfall application.

Four of the plots had volumetric moisture contents exceeding 20% before the dry run. Most plots, however, had values from 5 to 15%. These differences did not have a large impact on the data since the high value plots had a minimum of 25% or more cover.

No significant differences among regression slope values were found for ASL by application run versus the calculated cover factors (Table 2). Although the application run regression slopes were not significantly different, the following observations were made. There was a longer time to runoff and generally less runoff from the plots during the dry run. The regression slope change from the wet to very wet runs indicated that most of the loose soil particles on the surface had been removed during the dry and wet runs. Thus soil losses during the very wet run were being caused by raindrop detachment and surface scouring.

As the amount of bare soil increases, the Cover Factor increases. An increasing Cover Factor causes an increase in WSL as shown in Figure 1. With the exception of two data points, increasing percent cover above

Table 1. Range of variables for the erosion plots (n = 43).

	Range	
Variable	Low	High
Soil Parameters (0-50 mm)		
Very Fine Sand (%)	7.4	20.9
Sand (%)	25.3	83.5
Silt (%)	8.0	53.5
Clay (%)	8.4	25.4
Bulk Density (Mg/m^3)	0.8	1.4
Organic Matter (%)	2.0	5.1
Nomograph K[1]	0.016	0.050
Volumetric Moisture (%)[2]	2.8	24.3
Site Parameters		
Basal Cover (%)	0	100
Slope (%)	0.5	9.8
Age (Years)[3]	0	7

[1] From Wischmeier and Smith, 1978.

[2] Antecedent moisture before dry run.

[3] Vegetation age from seeding data.

Table 2. Correlation/regression results for ASL and WSL versus the cover factor.

	Regression Coefficients[1]		
Application Run	a	r^2	MSE
Dry	0.569A[2]	0.84	0.026
Wet	0.615A	0.86	0.026
Very Wet	0.578A	0.86	0.022
WSL	0.590A	0.90	0.015

[1] Form of equation: $y = ax$ where y = ASL or WSL and x = Cover Factor, r^2 = correlation coefficient, and MSE = mean square error. Intercepts were not significant at the $P = 0.10$ level.

[2] Values followed by the same letter are not significantly difference at the $P = 0.10$ level.

30% (Cover Factor less than 0.7) resulted in a tight data fit. This suggests a rapid decrease in WSL as cover is increased.

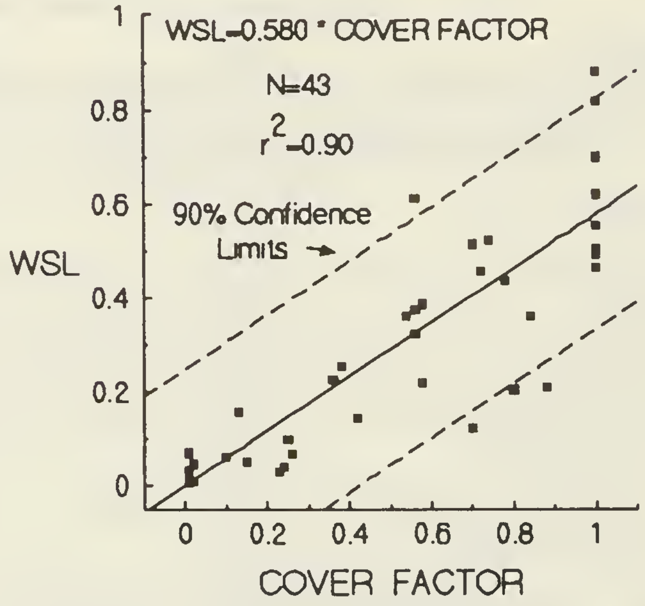

Figure 1. Relationship of WSL factor to the calculated cover factor for reclaimed soils.

Elimination of the bare plots from the data set resulted in two major effects on the regression equations. First, regression slopes, except for the very wet run data decreased by up to 3% (very wet, increased by 7%). Second, mean squared error values increased as much as 80% while all correlation coefficients decreased as much as 20%.

The differences in soil loss values between measured and estimated data for the 43 reclaimed plots were not large (Table 3). The largest variability found was for the dry run which can be attributed to factors such as the removal of loose soil particles from the soil surface and the duration and amount of runoff. Under -and over-estimated values were nearly equally distributed.

SUMMARY

Measured soil losses from simulated rainfall was used to relate adjusted soil loss factors to basal cover on reclaimed minelands. Sites were used that had a wide range of textures, slope gradients, and basal cover. Differences found among calculated regression slope values for the three simulated rainfall application runs were nonsignificant (P = 0.10). The regression slope value for weighted soil loss factor versus basal cover was similar to all the individual runs. Weighted soil loss factor values decreased linearly with increasing cover. Variability in the data points decreased once a minimum of 30% cover was achieved. The findings should help determine the percent cover needed after reclamation to minimize soil erosion on reclaimed land. The C factor needed in the USLE can be calculated and the equation in Figure 1 solved to determine the percent cover required.

Table 3. Summary of differences between measured and estimated soil losses from the reclaimed plots.[1]

Variable	Application Run Dry	 Wet	 Very Wet
	Under-Estimated Values		
Total Number	20	24	24
No. of Differences >1	3	2	1
Mean	0.636	0.291	0.223
Range:			
Low	0.024	0.014	0.003
High	3.087	1.305	1.234
	Over-Estimated Values		
Total Number	23	19	19
No. of Differences >1	6	1	1
Mean	0.649	0.320	0.360
Range:			
Low	0.025	0.030	0.059
High	1.968	1.665	2.024

[1] Difference (Mg/ha) = Measured - Estimated. Estimated values used field rainfall kinetic energy, USLE values for LS and K, and WSL values based on cover factor from Figure 1.

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PLANNING, REHABILITATION AND
TREATMENT OF DISTURBED LANDS

BILLINGS SYMPOSIUM, 1993

EROSION AND SEDIMENT CONTROL PRODUCT SELECTION FOR DRASTICALLY DISTURBED LANDS

William Agnew[1] and Marc S. Theisen[2]

ABSTRACT

Natural and synthetic erosion and sediment control materials have been used since the late 1950's and their use continues to expand at a rapid pace. Natural blankets and mulches tend to degrade over a short period of time and rely more on immediate vegetation response for erosion protection. Geosynthetic mattings and soil confinement systems endure the elements resulting in longer lasting erosion protection. Fiber roving systems can provide either short term or long term erosion protection and allows for flexibility of applications. In most cases, the long term solution to many erosion control challenges is the establishment of permanent vegetation. The dilemma faced by erosion control professionals is many sites require the use of erosion and sediment control materials to ensure a stable soil surface to facilitate seed germination and plant development.

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[1] Western Erosion Control Specialist, Synthetic Industries

[2] Product Manager Erosion Control, Synthetic Industries, Construction Products Division, 4019 Industry Drive, Chattanooga, TN 37416

INTRODUCTION

Natural and synthetic mats, blankets, fabrics, roving, soil confinement systems and geomatrices have been used for several decades to accomplish long term erosion protection in drainage channels, roadside ditches, steep slopes, benches, median strips, landscape projects, lake banks and other sensitive areas.

Temporary natural blankets and organic ground mulches tend to degrade in a short time and rely more on immediate vegetation response to allow long term erosion control protection.

Geosynthetic mats endure the elements providing long term protection to the soil surface and reinforced targeted vegetation.

In most cases, the long term solution to erosion control challenges is the establishment of permanent vegetation. Woody plants are often desirable for long term stability of a particular site. However, woody plants require time to develop sufficient size to control erosion adequately and the quick cover obtained by grasses and forbes is needed in the interim.

The dilemma faced by engineers, landscape architects and erosion control professionals is that many sites may require the use of natural or synthetic erosion-control material to ensure a stable soil surface to accommodate seed germination. Without soil surface stability, a project requiring vegetation establishment is destined to fail.

This article provides generic descriptions and specifications for three erosion control product types now on the market, specifically mats and blankets, roving and soil confinement systems.

Mats and Blankets

Natural (organic) mats and blankets generally are machine or hand woven products composed of wood excelsior, straw, coconut or other natural filaments and bound together with a tough photodegradable geosynthetic plastic mesh. The organic fiber mesh is evenly distributed over the entire area of the blanket. Natural blankets retain soil moisture, control surface temperature fluctuations of the soil, conform to the terrain, protect against sun burnout and break up rain drops to minimize erosion.

A high strength, flexible, woven geotextile erosion control mesh made of photodegradable polypropylene is also available. Initial high strength and gradual degradation make this product ideal for environmentally sensitive, bioengineering projects and is a low cost, proven, natural appearing alternative to jute, excelsior, straw or coconut blankets.

Geosynthetic mats typically are composed of a web of continuous or staple monofilaments thermally bound by heat fusion or stitched (mechanically bound) between nettings to provide dimensional stability. They are inert to chemicals normally encountered in a natural soil environment and stabilized against ultraviolet (UV) degradation.

Erosion control blankets and mats have ample thickness to provide cover and shading, resulting in improved soil moisture and enhanced seedling germination and are porous enough to allow a uniform and dense stand of vegetation to grow through the mat. Superior strength and performance as well as the mat's ability to remain functional for many years separates geosynthetic mats from degradable "high velocity" organic blankets.

Turf Reinforcement is a method or system by which the natural ability of plants to protect soil from erosion is enhanced through the use of geosynthetic materials. A flexible three dimensional matrix retains seeds and soil, stimulates seed germination, accelerates seedling development and most importantly, synergistically meshes with developing plant roots and shoots. In laboratory and field analyses, biotechnically reinforced systems have resisted flow rates in excess of four meters per second for durations of up to two days, providing twice the erosion protection of unreinforced vegetation (Carroll, Rodencal and Theisen, 1991). Such performance has resulted in the widespread practice of turf reinforcement as an alternative to concrete, riprap and other armor systems in the protection of open channels, drainage ditches, detention basins and steepened slopes.

Geosynthetic mattings generally fall into two categories: Turf Reinforcement Mats (TRM's) or Erosion Control Revegetation Mats (ECRM's). Higher strength TRM's provide sufficient thickness and void space to permit soil filling/retention and the development of vegetation within the matrix. TRM's are installed first, then seeded and filled with soil. Seeded prior to installation, ECRM's are denser, lower profile mats designed to provide long term ground cover and erosion protection. By their nature of installation TRM's can be expected to provide more vegetative entanglement and long term performance than ECRM's. However, denser ECRM's may provide superior temporary erosion protection. Geosynthetic mattings occupy one of the fastest growing niches of the erosion and sediment control industry.

Manufacturer recommended procedures should be followed to install any erosion control mat or blanket. Whether slope or channel, the site preparation prior to any erosion control blanket or mat installation must be shaped to design specifications and dressed to be free of soil clods, clumps, rocks or vehicle imprints of any significant size that would prevent the erosion control material from lying flush with the soil surface.

Fiber Roving Systems

Roving is a method of erosion control used in drainages, slopes, etc., besides the traditional choices of blankets, mats, mulches or other erosion control products. Fiberglass and polypropylene materials are used to form a continuous strand for erosion control protection.

Fiberglass roving is a material formed from fibers drawn from molten glass. There fibers are gathered in strands to form a single ribbon.

Polypropylene roving is replacing fiberglass in the erosion control marketplace. This roving is formed from continuous strands of fibrillated polypropylene strands. The roving is wound onto cylindrical packages so that the material can be fed continuously from the outside of the package. The advantage of polypropylene is that it does not require the safety equipment (gloves and mask to prevent skin and lung irritation) necessary for fiberglass roving applications.

The fiber roving system is unique in the industry because of the flexibility of application, allowing for any width or thickness of material to be applied. Other erosion control materials, such as blankets or mats, require the user to apply the width of material and/or thickness that is supplied. Since fiberglass and UV treated polypropylene are resistant to photo and biodegradation, they are a long lasting erosion control material. Conversely, natural polypropylene (non-UV treated) roving provides high strength to resist moderate flow velocities and biodegrades similar to organic blankets.

Fiber roving systems work just like any other erosion control material in that it serves as a protection against soil loss caused by either rain fall, wind or water movement while vegetation is becoming established. As the vegetation becomes established, roving facilitates root reinforcement and aids in permanent vegetation establishment.

The installation of fiber roving in either a drainageway or on a steep side slope is a relatively simple procedure. The equipment needed for installation is standard highway maintenance equipment that is easily acquired by an erosion control contractor. Equipment needed includes an air compressor capable of producing 100 cfm at 100 psi, and either an emulsified asphalt tank or some sort of tank with a pump attached that can pump emulsified asphalt or other appropriate tackifier through a handgun.

After selecting the area to be treated, the soil should be prepared either mechanically or manually so seed, fertilizer and/or lime may be applied. The selection of a seed mix should be predicated on the native species of the area with the goal of obtaining strong root structures.

The erosion control roving is applied using a special nozzle connected to the air compressor. As the roving feeds into the nozzle, compressed air, regulated at the nozzle to an approximate pressure of 100 psi, propels the roving out of the nozzle, separating the strands into individual fibers.

The roving is spread over the area to form a random mat of continuous fibers. As previously mentioned, one of the plusses of using a roving is the ability of the applicator to vary the width of the "blanket of roving" as well as the thickness to match up with areas of higher erosion potential (Agnew, 1991). After completing the application a tack coat of emulsified asphalt or other appropriate tackifier is applied to assure adhesion of the fibers to one another and to the soil.

It has been suggested emulsified asphalt is a poor environmental choice, not to mention the "black" color is aesthetically unappealing. Environmentally, emulsified asphalt has been shown to be very stable when enough time is allowed for the material to "set up" before any precipitation event.

There is no defense for the aesthetics except in areas where germination and vegetation growth occur within a few weeks and the visual effect is limited. In areas where the growing season is short, the "black" coloring is judged to be beneficial to assisting in the germination process by collecting heat.

Several types of tackifiers have been tested by the authors as well as other users, but so far nothing tested has worked as successfully as the emulsified asphalt. Additional testing on other viscous, non-water soluable tackifiers is ongoing with promising results for future applications.

The polypropylene roving is typically applied at a rate of .15 to .25 lb/sq. yd. Again, the application rate depends on the discretion of the applicator to meet the erosion control potential of the project. The emulsified asphalt is applied at a rate of .25 to .35 gal/sq. yd. Applied cost of erosion control roving depends on application rate, the scope of the project and the area in which the project is geographically located.

Soil Confinement Systems

Soil confinement systems generally consist of a series of honeycomb like cells formed into a spreadable sheet or blanket. Sheets of the material are anchored and filled with soil, creating a solid, pavement-like surface in areas of poor soil stability. The products generally are made from a high density polyethylene or nonwoven polyester material.

Soil confinement systems are designed to address engineering and excavation projects such as earth and slope surface reinforcement, highway embankments, headwalls and wingwalls, pipeline and culvert installations, highway bed containment, and excessive drainage slopes, to mention a few. Similar to erosion control blankets, mats and roving, soil confinement systems can provide a suitable soil surface for vegetation establishment.

The soil confinement geomatrix will hold varying granular materials such as sand, pea gravel and other aggregate that require containment as a support material. It's most appropriate for dry or sterile areas where little or no vegetation can be expected.

Performance of Erosion Control Materials

Several test procedures have been proposed to quantify performance of erosion control materials. Initial concern for vegetated systems is temporary erosion protection prior to and during seed germination and seedling development. Typically, this level of performance is measured by the material's ability to minimize soil loss when subjected to various flow rates and/or rainfall amounts. Temporary erosion protection is important but the long term goal of any vegetated erosion control matrix is to provide permanent erosion protection via permanent vegetation and subsequent root reinforcement. The more rapidly vegetation becomes established the more rapidly long term erosion control may be accomplished. Thus, the material's ability to facilitate vegetative establishment is equally important. Too much emphasis on an erosion control product's temporary protection may inhibit the growth of newly emerging seedlings.

Perhaps the most critical parameter in an engineering design is flow resistance before, during and long after vegetative establishment. Some erosion control materials may be washed away before the vegetation takes hold while other may temporarily exhibit excellent flow resistance only to lose their effectiveness as they degrade or decompose over time. Specifiers must take into account immediate and long term flow resistance based upon longevity of the material when designing grassed slopes and waterways.

Two basic design concepts are used to evaluate and define a channel configuration that will perform within accepted limits of stability. These methods are defined as the permissible velocity approach and the permissible tractive force (shear stress) approach. Under the permissible velocity approach the channel is assumed stable if the adopted velocity is lower than the maximum permissible velocity. The tractive force (boundary shear stress) approach focuses on stresses developed at the interface between flowing water and the materials forming the channel boundary (Chen and Cotton, 1988).

The permissible velocity approach uses Manning's Equation where with given depth of flow, D, the mean velocity may be calculated as:

$$V = 1.49 R 2/3 S 1/2 /n$$

where V = average velocity in the cross section;
n = Manning's roughness coefficient;
R = hydraulic radius, equal to the cross-sectional area, A, divided by the wetted perimeter, P, and
S = friction slope of the channel, approximated by the average bed slope for uniform flow conditions.

The tractive force approach uses a simplified shear stress analysis which is equal to:

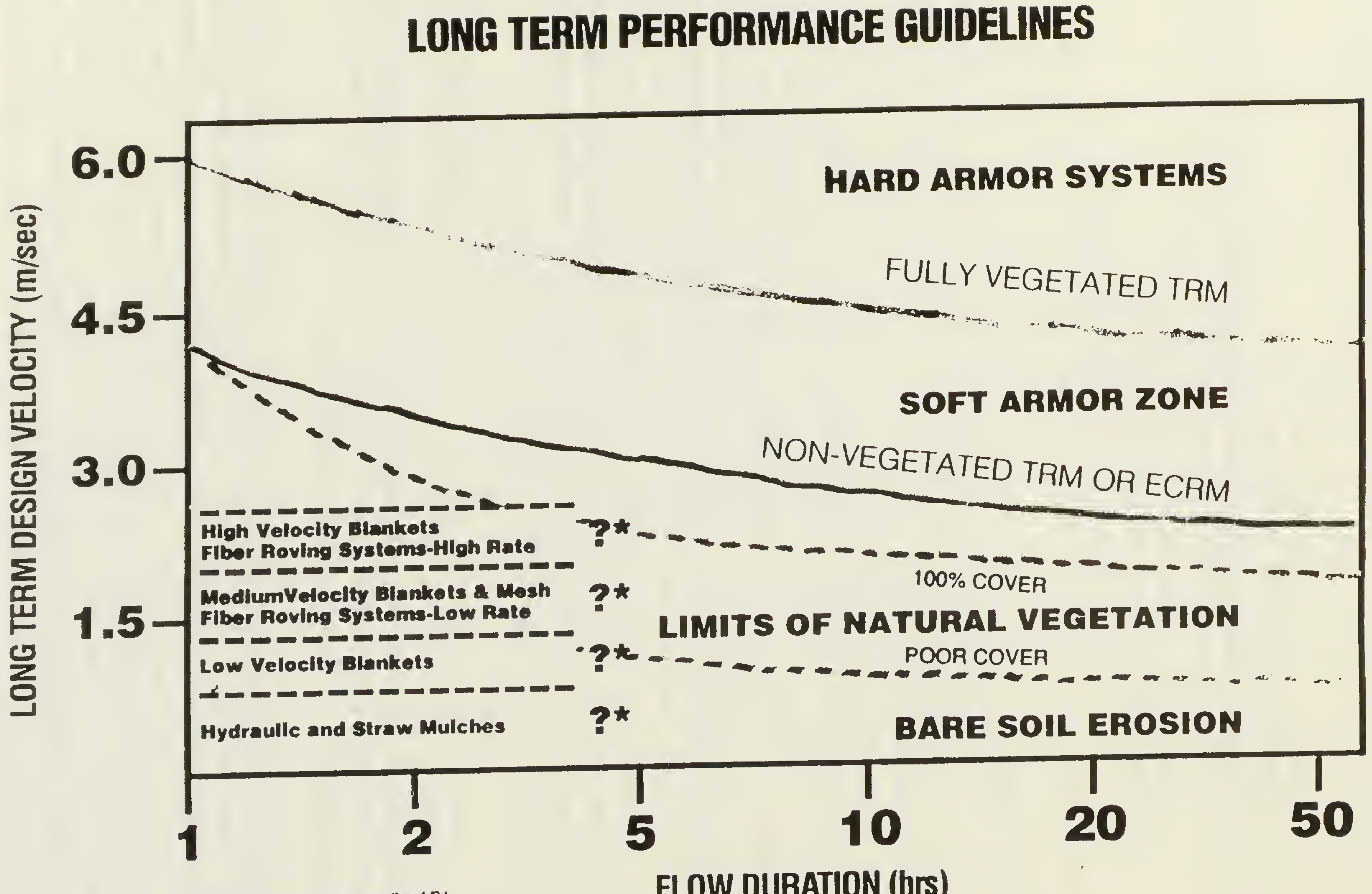
$$T = yDS$$

where T = tractive force;
y = unit weight of water;
D = maximum depth of flow;
S = average bed slope or energy slope.

Design criteria based on flow velocity may be limited because maximum velocities vary widely with channel length (L), shape (R), and roughness coefficients (n). In reality it is the force developed by the flow, not the flow velocity itself, that challenges the performance of erosion control systems. Tractive forces caused by flowing water over the ground surface create shear stresses which can be used as a design parameter independent of channel shape and roughness. Moreover, the higher stresses developed in channel bends or other changes in stream channel geometry can be quantified by simplified shear stress calculations, providing a higher level of design confidence than otherwise possible (Chen and Cotton, 1988).

Critical shear stress determinations are meant to be used with velocity calculations for prescreening of channel lining designs. Manning's Equation remains the primary hydraulic research and design tool. However, as everyday practice has determined, a simplified screening criteria such as maximum shear stress is necessary to ensure properly engineered design of channel lining erosion control systems (Theisen, 1992).

Maximum permissible velocities for vegetative techniques are illustrated under vegetated and non-vegetated conditions (Figure 1). The figure graph illustrates maximum permissible limiting velocities versus flow duration compiled from existing published evaluations of geosynthetic mattings and classic erosion control studies. Thus a designer will have performance guidelines from the time a material is installed to when it becomes fully vegetated. As additional data becomes available from field and laboratory analysis perhaps a design guide chart utilizing maximum permissible shear stress may be developed. The reader must be cautioned, velocity and tractive force are not directly proportional. Under certain conditions, a decrease in velocity may increase depth of flow, thereby increasing shear stress.



*No known published data exceeding 1/2 hr

FIGURE 1

Conclusion

Regardless of which erosion control material (blankets, mats, roving, soil confinement systems and others) is used for a specific project, if vegetation is desired, the final product, if installed properly, will provide significant long term erosion control protection. Consult with the manufacturer and/or their representative to ensure proper material selection and installation will result in the successful implementation of erosion control projects at the least cost to you.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

UNEXPECTED IMPACTS TO UNMINED AQUIFERS NEAR COAL MINES

Jon Reiten[1] and John Wheaton[1]

ABSTRACT

Coal beds targeted for mining commonly serve as one of the most dependable water resources for the traditional agricultural based economy in southeastern Montana. As a result of the dual importance of the coal-bed aquifers, impacts to the hydrologic system caused by mining has been a focus of research. More than twenty years of monitoring has resulted in the documentation of expected impacts, as well as some unexpected ones. In general, hydrostatic pressures following mining are approaching pre-mining conditions. Water-quality degradation appears to be the most persistent problem in the aftermath of large-scale strip mining, and the duration of this problem has yet to be clearly identified. Dissolved-solids concentrations are shown to increase 200 percent or more during resaturation with strong increases in dissolved concentrations of sodium, calcium, magnesium, and sulphate ions.

Observed impacts to the ground-water system are for the most part consistent with predicted impacts; however, documented water-level declines in unmined aquifers are an exception to earlier predictions. These drawdowns appear to be caused by inadequately sealed exploration holes forming hydraulic connections between mined and unmined aquifers. Unsealed boreholes would allow migration of poor-quality spoils water into the unmined aquifers.

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[1] Montana Bureau of Mines and Geology, Billings, Montana

INTRODUCTION

Predicting hydrologic impacts to the ground-water system near surface coal mines is critical to the protection of water resources in southeastern Montana. The Montana portion of the Fort Union coal region (Figure 1) contains over 32 billion tons of lignite and subbituminous coal. The coal beds are the most widely used and dependable aquifers in this region because of their generally fractured nature and lateral continuity.

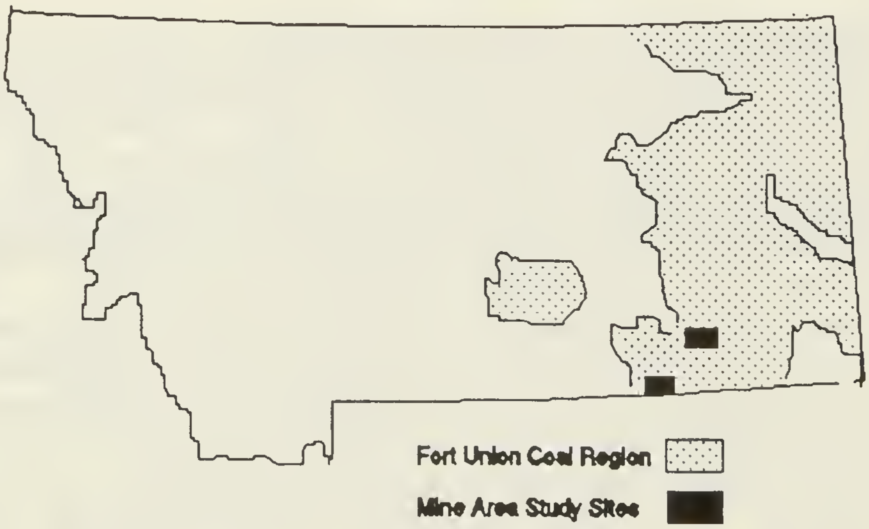


Figure 1. Locations of Fort Union Coal Region and mine area study sites.

Over two decades of observations have produced an accurate picture of the hydrologic impacts thus far near coal strip mines in Montana. In general the observations follow the predicted impacts developed in the 1970's by the Montana Bureau of Mines and Geology (MBMG), the U.S. Geological Survey (USGS), state regulators, and mine company hydrologists (USGS, 1974; Van Voast and others, 1977; USGS, 1979; Dollhopf and others, 1981; Van Voast and Reiten, 1988). For the most part, post-mining ground-water levels appear to be approaching pre-mining levels. Recovery to pre-mine water quality is not evident.

A potentially significant exception to earlier predicted impacts has come to light as a result of the long-term hydrologic monitoring conducted by the MBMG (Van Voast and Reiten, 1988). Significant water-level declines in deeper unmined aquifers have been documented near Colstrip and Decker. There is very little potential for sufficient vertical flow through the intervening layers of low permeability clay and silt. A possible explanation for the observed drawdowns is that unplugged or poorly plugged drill holes may be causing significant drawdowns in mining areas.

PREDICTED IMPACTS

Water-level and water-quality predicted impacts were developed in the 1970's based on hydrogeologic investigations that described the ground-water systems near planned coal mines. The predicted impacts were assumed to be relevant to other potential mine sites in this and other mining regions.

Predicted Impacts to Aquifer Water Levels

A typical measure of the quantity of water in an aquifer is the water level or the potentiometric level in observation wells. Figure 2a depicts premining conditions with the coal-bed aquifer under steady state conditions. Since coal beds are both aquifers and mining targets, both the aquifer matrix and aquifer water will be removed during surface coal mining. Consequently, a logical predicted impact is that, during mining, water levels will decline in and adjacent to mined areas (Figure 2b), and that the areas of influence can be estimated with reasonable confidence.

Recharge to the mine spoils is from lateral flow through adjacent unmined aquifers and vertical infiltration of rainfall and snowmelt. Consequently, as long as regional precipitation occurs at rates similar to historic conditions, water quantity, and therefore hydrostatic pressures, will recover to approximately pre-mine conditions after dewatering of active mine pits ceases (Figure 2c). Upward flow from deeper aquifers has traditionally been accepted as leakage through poorly permeable underlying mine floors and has been given little significance.

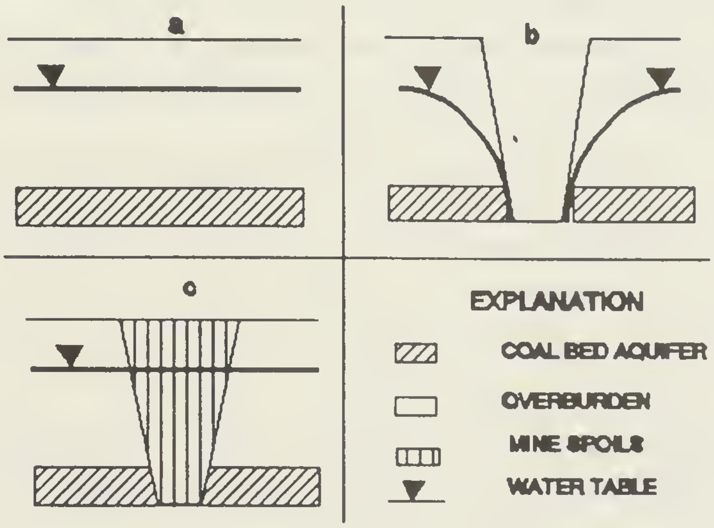


Figure 2. Typical history of hydraulic-head fluctuations in a coal-bed aquifer starting with pre-mining background conditions (2a), progressing through mining induced drawdowns (2b), and ending with resaturated spoils approaching pre-mining conditions (2c).

Impacts to the ground-water system are expected to be restricted to the mined out coal bed, and aquifers in the overburden. Water-level declines are expected to occur in portions of these aquifers that are adjacent to and within 1 to 2 miles of the active mine.

Predicted Impacts to Aquifer Water Quality

Water quality has been predicted to be degraded in the spoils and aquifers downgradient of the spoils. The post-mining range of total dissolved solids has been predicted to increase by at least as much as two or more times. The chemical nature of water in the spoils aquifers would be similar to water in undisturbed aquifers dominated by ions of sodium, calcium, magnesium, and sulfate. As salts are flushed out of the spoils, water quality would eventually improve to pre-mining conditions.

OBSERVED IMPACTS

Impacts to the hydrologic system near surface coal mines have been documented by over 20 years of ground-water monitoring on a network of over 200 wells in the Colstrip and Decker areas. Hydrographs depicting water-level fluctuations show trends that can be explained by mining-related activities, natural background conditions, and the activities of people. Repeated water-quality analyses at selected observation wells show a dynamic chemistry in aquifers directly impacted by mining. Aquifers not impacted by mining typically show significantly less variability in aquifer water chemistry.

Observed Impacts to Aquifer Water Levels

The most obvious impact to the ground-water system directly caused by mining is the water level declines that occur in disturbed aquifers. Mining of the Rosebud coal bed, for example, has caused water-level declines in unmined portions (Figure 3). These water-level declines are consistent with predicted impacts derived from research conducted in the 1970's.

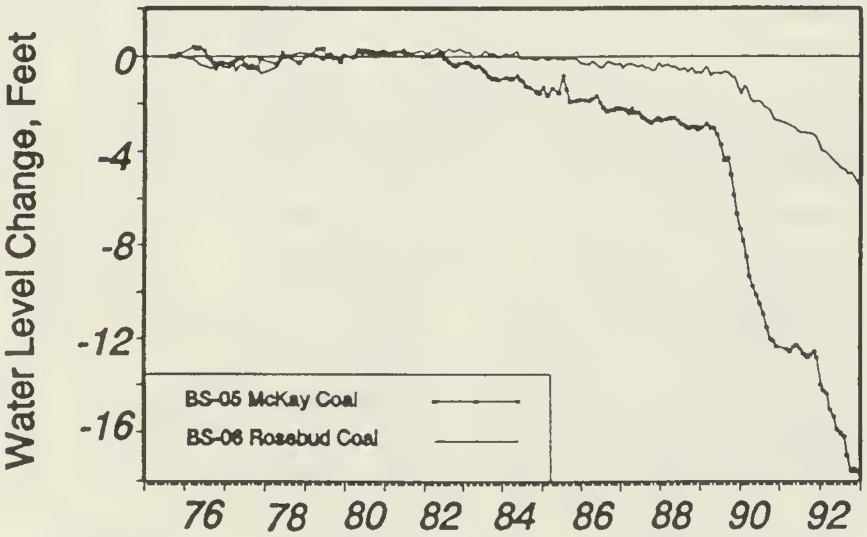


Figure 3. Hydrograph depicting drawdowns caused by mining the Rosebud coal bed.

Water-level declines in unmined portions of coal and overburden aquifers have been observed adjacent to nearly all active pits. The magnitude of these declines is controlled by the hydrostatic pressure, hydraulic conductivity, and storage coefficient of the aquifer. As pits enlarge and stay open for an extended period of time, more water is removed from storage causing increasing water-level declines. Lateral extent of the affects varies directly with aquifer transmissivity and inversely with storage coefficient (Freeze and Cherry, 1979). Vertical extent varies directly with storage coefficient and inversely with transmissivity. Consequently, the interplay of ground-water discharge caused by mining and pit dewatering, and the intrinsic aquifer characteristics, controls the observed water-level declines in aquifers adjacent to mining.

Mining in the Decker area has created a potentiometric depression more than 15 miles long and 5 miles wide with drawdowns ranging between 10 and 70 feet. Drawdowns in the Colstrip area are restricted to within 1 to 2 miles of active mine pits and are generally less than those in the Decker area.

Recharge to spoils aquifers has occurred in all areas currently being monitored. Examples of resaturation are shown in Figure 4 where infiltration of recharge water (EPA-12) and water from adjacent unmined coal beds (BS-29) appears to be recharging the spoils aquifer. Several years are required before the spoils water level reaches a steady state.

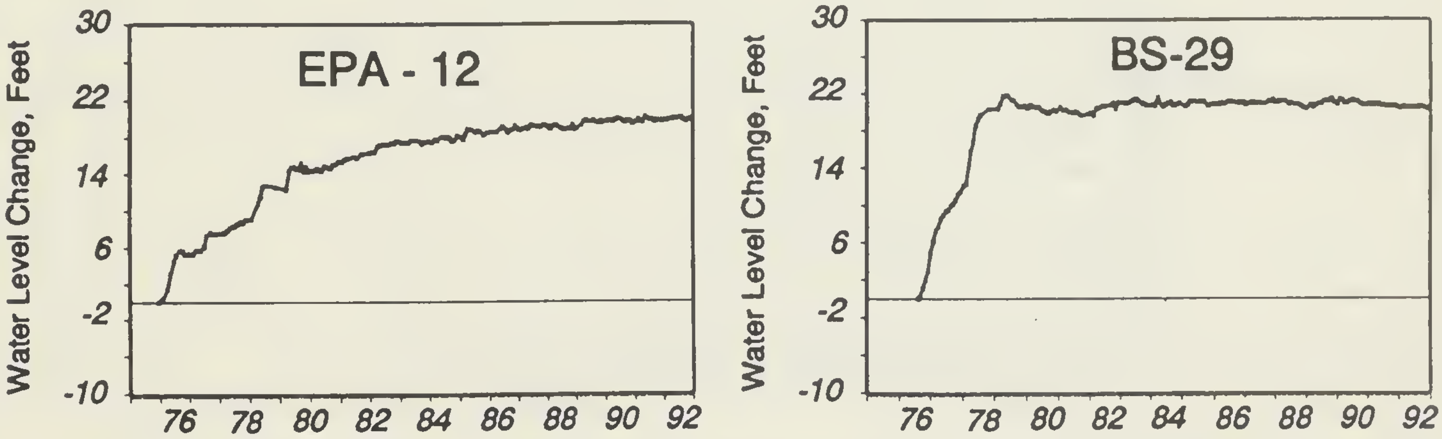
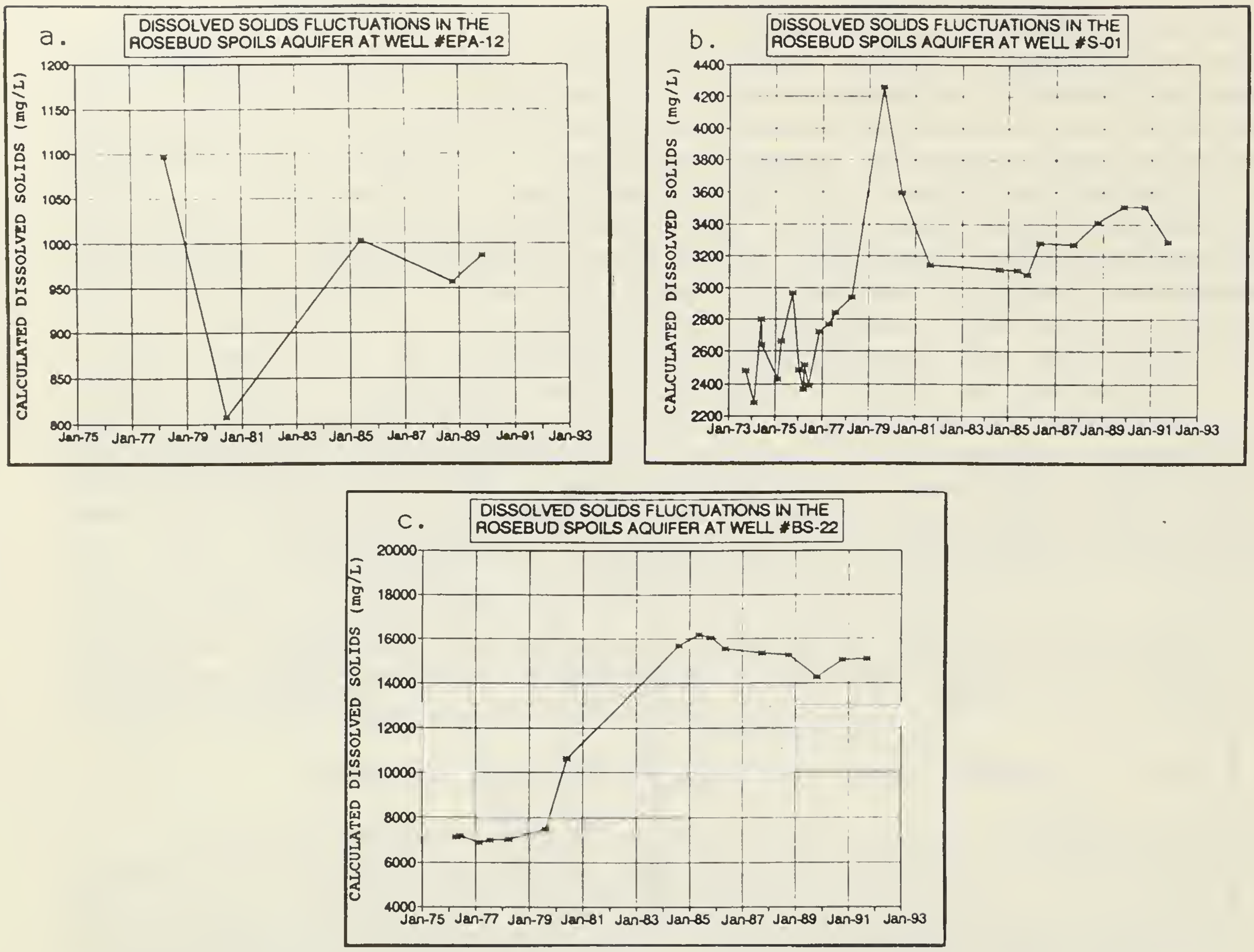


Figure 4. Hydrograph depicting resaturation of mine spoils at EPA-12 and BS-29.

Observed Impacts to Aquifer Water Quality

Water-quality trends in the spoils are very diverse and appear to be largely dependent on site specific ground-water flow and availability of soluble salts. In general, dissolved-solids concentrations in spoils are increased over 200 percent of concentrations in unmined aquifers. Most of the increased dissolved solids are the result of increases in highly soluble salts containing sodium, calcium, magnesium, and sulphate ions. At well EPA-12 active flushing caused by infiltration of ponded surface water on top of mine spoils has removed much of the available salts in this area (Figure 5a). In contrast, slower flushing rates at well S-01 indicate dissolved solids varying directly with water levels or increasing with time (Figure 5b). Probable worst case conditions are shown by water-quality trends at BS-22 where localized conditions have resulted in very high concentrations of salts in the spoils aquifer (Figure 5c). At some wells in mine spoils, water-quality trends are towards decreasing concentrations of dissolved solids, indicating that recharge moving through the spoils is flushing salts from the spoils aquifer.


DISSOLVED SOLIDS FLUCTUATIONS IN THE ROSEBUD SPOILS AQUIFER AT WELL #EPA-12

DISSOLVED SOLIDS FLUCTUATIONS IN THE ROSEBUD SPOILS AQUIFER AT WELL #S-01

DISSOLVED SOLIDS FLUCTUATIONS IN THE ROSEBUD SPOILS AQUIFER AT WELL #BS-22

Figure 5. Dissolved-solids concentrations show a variety of trends in spoils aquifers.

Dissolved-solids concentrations show a variety of trends in spoils aquifers. Spoils water moving under normal gradients has been predicted to degrade water quality in downgradient alluvial and bedrock aquifers. The predicted offsite degradation has not been clearly documented, although some has been observed. The ultimate destination of the salt load being carried by the spoils water must be off site either to adjacent or deeper aquifers.

UNEXPECTED HYDROLOGIC IMPACTS

As the Rosebud coal is being removed, not only are aquifer water levels declining in unmined portions of this coal bed, but even greater declines have been observed in the deeper McKay coal aquifer (Figure 3). The interval separating these coal beds ranges from 20 to 50 feet thick, consisting largely of siltstone, clay, and shale. These fine-grained sediments have very low permeabilities and will not readily transmit water. Several minor faults have been mapped in this area; but faults have been found to be barriers to flow in similar Fort Union sediments. Consequently, it is difficult to explain a hydraulic connection through the intervening confining beds that would allow the observed drawdowns to occur. Similar responses are being documented in the Decker area.

The fact that water levels in unmined aquifers are declining, indicates that a hydrologic connection exists through the intervening confining interval. It appears that unplugged or poorly plugged exploration holes are hydraulically connecting mined intervals with deeper unmined aquifers. The cone of depression caused by removing the target coal bed is being transmitted through the exploration holes, in effect short-circuiting the natural flow barriers. The water-level response within the unmined coal is greater than in the mined coal at similar distances from active mining at some monitor sites. The different magnitude of water-level responses in different aquifers is probably the result of differing characteristics between the aquifers. Specifically, pre-mining hydrostatic pressure, storage coefficient and hydraulic conductivity are major factors that affect aquifer water-level responses. Additional factors include distance and degree of hydraulic connection between the mine pit and the aquifer.

Large numbers of holes are drilled while exploring and mapping coal resources. Figure 6 depicts drill-hole density in an area of active mining, as well as the area of coal mined during a four year period. Drill-hole density ranges up to 40 holes per million square feet (23 acres). Most of these holes were reportedly plugged according to the regulations in place at the time of drilling, which included plugging with cuttings, bentonite slurries, or bentonite chips. Apparently, at least some of these plugging techniques did not adequately seal many of the holes. Figure 7 shows a cross section of drawdowns measured after 4 years of mining. The section is offset from the mine pits, trending approximately parallel to both the mine and to the ground-water gradient.

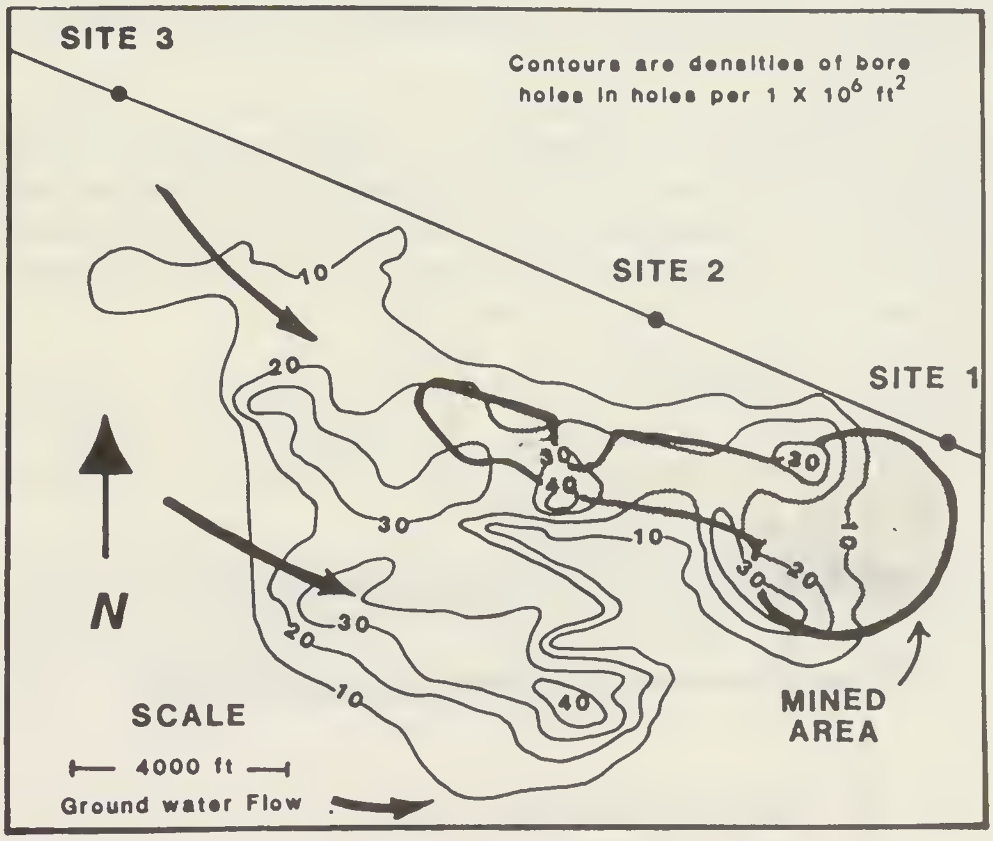


Figure 6. The location and density of exploration drilling reflect coal outcrops, topography and property boundaries.

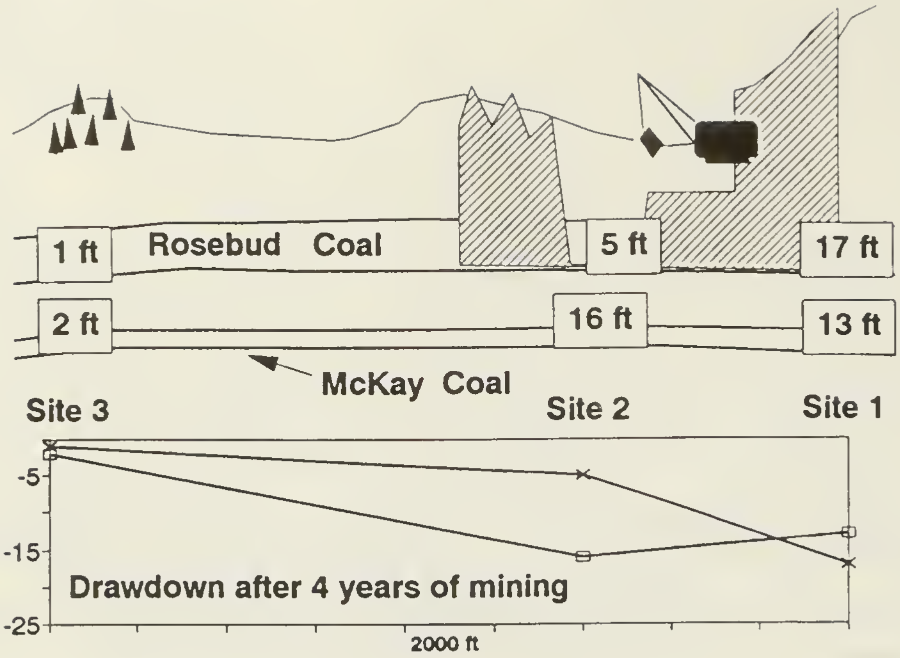


Figure 7. A generalized cross-section near an area where the Rosebud coal is being mined, showing significant drawdown in the Rosebud aquifer as well as in the unmined McKay aquifer. Location of cross-section is shown on Figure 6.

Drawdowns at site 1, which is located closest to mine pits, are 17 feet in the Rosebud aquifer and 13 feet in the McKay aquifer. Water levels in both aquifers at site 1 are now recovering as mine pits are being backfilled and distance to dewatered pits increases. Drawdowns at site 2 and site 3 are reversed in magnitude with greater declines in the unmined McKay aquifer than in the Rosebud aquifer. Water levels in both aquifers at these sites are continuing to decline as the cone of depression expands.

The relationship of exploration-hole drilling to later mining is illustrated in Figure 8. Exploration holes typically penetrate through the McKay coal into deeper sandstone and coal beds. Apparently a number of the exploration holes are not adequately sealed, allowing drawdowns to be transmitted through relatively impermeable confining beds.

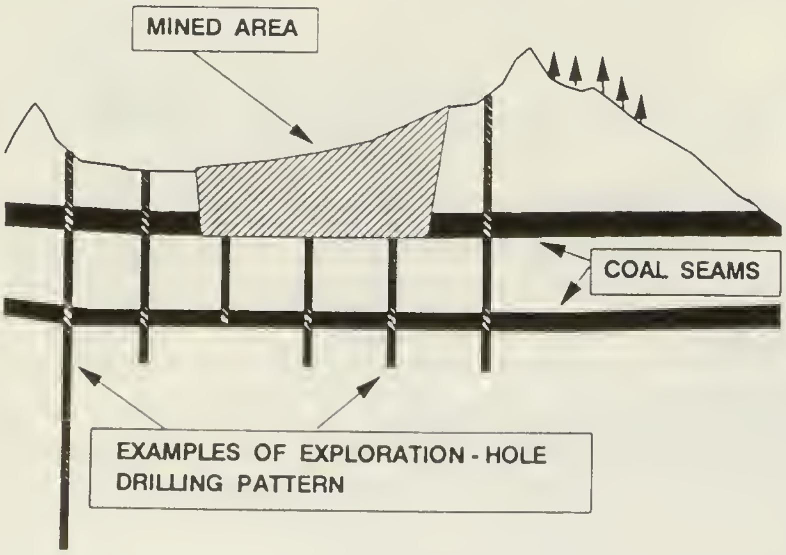


Figure 8. The relationship of exploration-hole drilling to coal mining.

Water Quality Implications

The unsealed drill holes will act as conduits for the migration of poor-quality spoils water to deeper, and otherwise unimpacted aquifers. While water quality degradation is likely in the deeper aquifers, it has not yet been documented. The travel time for migration of the salt loads cannot be calculated because the flow paths are so complex.

Aquifers in sandstone deposits underlying the McKay coal have been commonly suggested as a likely source for replacing stock wells that were mined out. Natural hydraulic gradients are downward in most of the mining areas. As mine spoils are backfilled and resaturated, lower quality water may potentially move under normal hydrologic gradients through poorly sealed drill holes into the deeper aquifers. The unexpected impacts to unmined aquifers is likely to be more significant than anticipated.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

BENTONITE AS A HOLE ABANDONMENT AND ANNULAR SEAL MATERIAL

by

Steve Regele
Dave Clark
Bob Bohman

John Wheaton
JoJean Lyford

ABSTRACT

A large number of boreholes are drilled each year for water supplies, ground-water monitoring, mineral and seismic exploration, and engineering purposes. Any borehole is a potential channel for impacts to ground-water. If left open, or improperly sealed, these holes can become an avenue for ground-water contamination, lowering of water levels, and ground stability problems. Current law requires the protection of ground-water and other natural resources. Documentation of valid successes and failures of borehole sealing materials and material emplacement strategies is vital. The choice of a sealing material must be based on an understanding of the down-hole conditions, and the strengths and weaknesses of the available treatment types. Bentonite hole plugging materials with a high bentonite solids content provide a stable hole abandonment column, and a more competent seal than low solids bentonite slurries. Low solids slurries of bentonite do not provide either a physically stable or hydrostatically competent seal.

Steve Regele, Dave Clark and Bob Bohman, MT Department of State Lands, Airport Industrial Park, Billings, MT 59105-1978

John Wheaton and JoJean Lyford, MT Bureau of Mines and Geology, 1500 N. 30th, EMC Box 112, Billings, MT 59101

INTRODUCTION

Protection and wise utilization of water and other natural resources is of paramount importance. Degradation of water quality and diminution of water quantity occurs in numerous ways. One common avenue for contamination and diminution is provided by bore holes. Bore holes are created as a result of water well construction, mineral exploration, and geophysical data acquisition. If these holes are not adequately plugged when they no longer serve a useful purpose, or if production zones are not effectively sealed from all other zones, water and other natural resources can be degraded or lost, and data can be invalidated.

Specific purposes for grouting a bore hole include: 1) providing an effective hydrologic sealant in abandoned bore holes (prevent intermingling of aquifers, aquifer contamination, draining of aquifers, loss of hydrostatic pressure); 2) isolating production or monitored zones from all other zones in water and monitor wells; 3) preventing the escape of oil, natural gas, etc.; and 4) minimizing ground surface subsidence.

A wide variety of grout materials and grouting methods are used in efforts to plug bore holes, and as a means of sealing the space between a bore hole wall and well casings. For our purposes, we define "grout" as an inorganic slurry or a solid material with low permeability to water, that is pumped or gravity fed into a bore hole to specific locations for specific purposes. The term includes: bentonite slurries (measured in pounds per gallon of bentonite in water); dry bentonite powder, granules, pellets or chips; drill cuttings (generally restricted to clay cuttings); cement (generally a mixture of 94 lbs. of dry Portland cement per six or less gallons of water); concrete (a mixture of Portland cement, water and sand or gravel, etc.); other inorganic materials; various combinations of the above materials.

Bentonite (montmorillonite) grouts are the central focus of our research. Other materials were included for comparative purposes. Bentonite has the appealing quality of being able to swell from 10-15 times its dry volume, thereby conforming to at least some irregularities encountered in bore hole walls. It is possible for bentonite to hydrate and swell, desiccate, rehydrate and swell, an infinite number of times. Also, bentonite seals are flexible, and may reseal if perturbations disrupt an initial seal. There are some limitations on the performance of bentonite grouts. For instance, slurries of bentonite can flow into fractures or porous formations thereby providing no seal, and chemical characteristics of make-up water or groundwater may adversely affect the performance of bentonite.

In an effort to better define and document some aspects of the utility and limitations of various bore hole sealing materials (grout) and grouting techniques, the Montana Department of State Lands and the Montana Bureau of Mines and Geology are conducting a joint research project.

The ongoing analysis of different grouts includes: 1) pumping and gravity methods available for emplacing several grouts; 2) simulated effects of water chemistry and insitu pressures (hydrostatic, gas, etc.) on these grout materials and emplacement techniques; 3) the effect of seismic activity on the performance of the grouting materials; 4) a limited cost/benefit analysis of some techniques and materials.

Trials were replicated whenever possible to allow for statistical analysis. Reasonable measures were taken to insure consistency of tested materials. Commercial products were obtained from a variety of manufacturers. No endorsement or critique per commercial products is provided as a result of our study.

While portions of the study are still being completed, an overview of findings regarding the quality of the seal of some different grouts is reported here.

RESULTS AND DISCUSSION

Shallow Hole Trials

Six inch diameter holes were drilled to approximate depths of 25 feet in three different lithologies. Holes were drilled in: 1) "clinker" (baked sediments of the Tongue River Member of the Fort Union Formation) at the Decker Mine near Decker, Montana; 2) sandstone (Eagle formation) northeast of Billings, Montana; and 3) shale (Bearpaw formation) south of Billings. These sites were chosen to approximate respective worst, middle and best case drilling and hole abandonment conditions typically found in eastern Montana. The "clinker" unit was comprised of highly fractured porcellanite, sandstone and siltstone (fractures of one-inch plus, common throughout the column). The sandstone unit appeared consistent from top to bottom (no evident fractures or notable differences in overall porosity) and is a medium grained sandstone unit with moderate to low porosity. The shale site was selected as likely to represent low to very low porosity. This unit also appeared to be consistent from top to bottom. No aquifers were encountered in any of the drilling. While some moisture was noted in the formations, none of the holes were saturated - i.e. no free water in any holes. The depth of unconsolidated material (soil) overlying the lithosphere at all sites was less than one foot.

Five grouting treatments were applied to these holes: 1) An 8.6 lb/gal slurry (commercial grade bentonite powder and water); 2) a 9+ lb/gal slurry (commercial grade bentonite powder and water); 3) dry bentonite chips (commercial grade, average size 3/4 inch, ± 1/4 inch); 4) pit run or "low grade" dry bentonite (unsorted, fine through coarse material, average size about 1/2 inch, ± 3/8 inch containing some silt and rock fragments); and 5) a dry mixture of 25% bentonite (commercial grade) and 75% gravel (average size about 3/8 inch, ± 1/8 inch). Slurries were pumped into place, dry materials were gravity fed by pouring into place. Measurements were taken from a stationary measuring point directly above each bore hole opening. Figures 1 through 3 depict the generalized results of the trials.

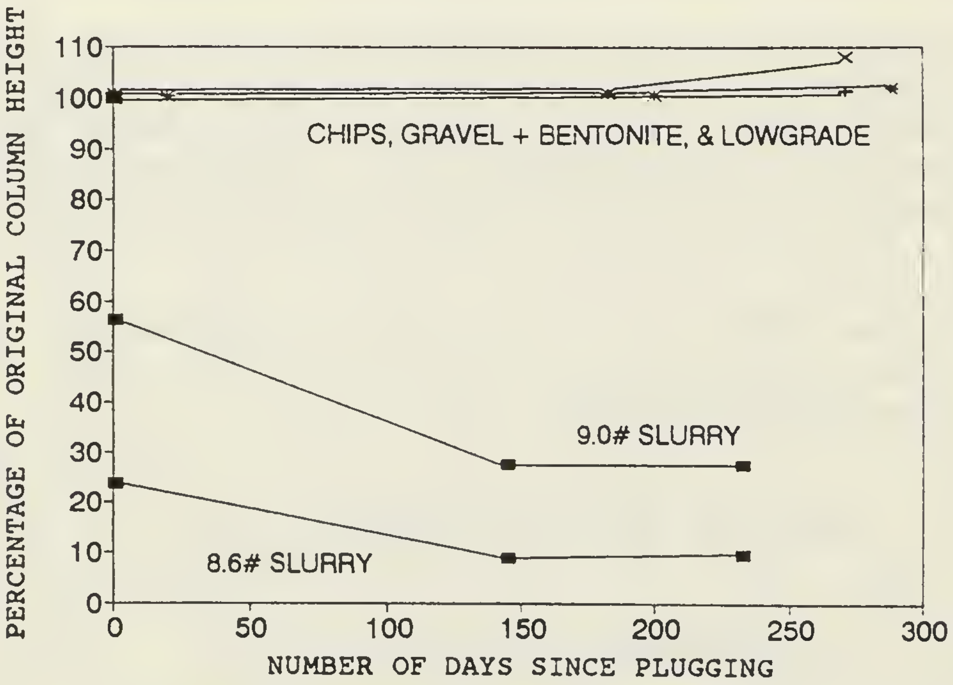

FIGURE 1. Clinker holes: mean change in plug column heights with time.

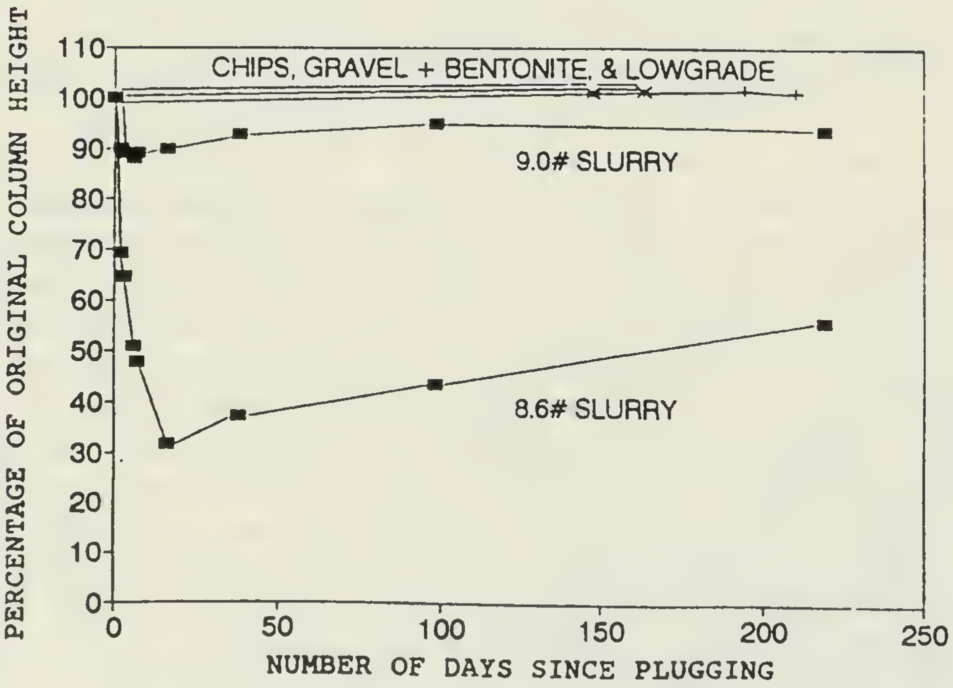

FIGURE 2. Sandstone holes: mean change in plug column heights with time.

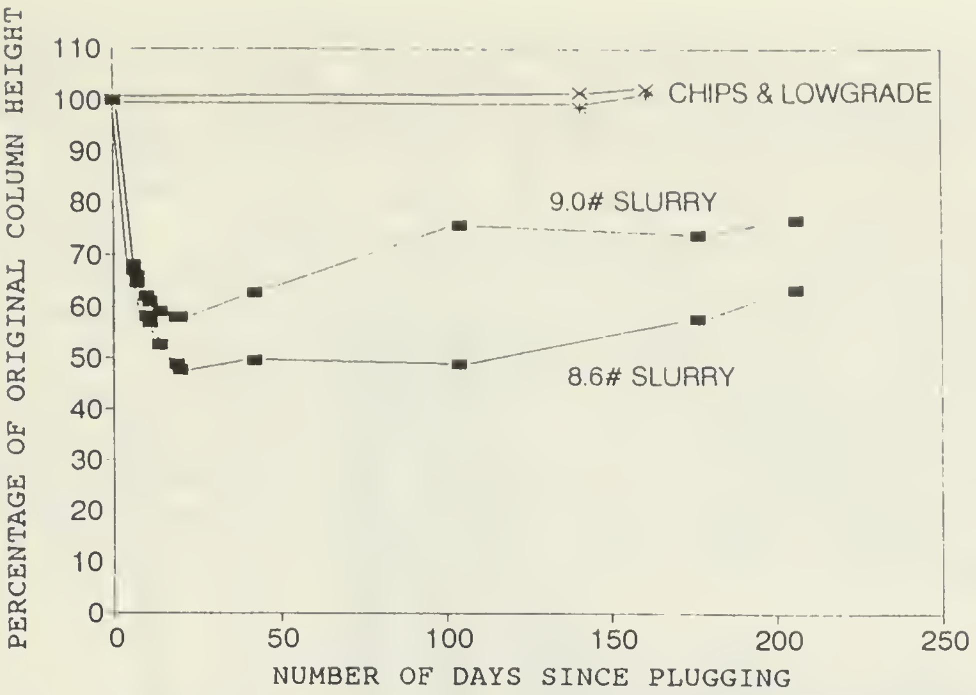

FIGURE 3. Shale holes: mean change in plug column heights with time.

The physical stability of the commercial grade bentonite chip, low grade bentonite chip, and the bentonite/gravel treatments (≈80% solids/unit volume) were similar regardless of site lithology. Each of these treatments either remained stationary at the installed level (100% fill), or rose slightly in the holes due to swelling of the bentonite.

Settling of slurry plug treatments (≤20% solids/unit volume) occurred in all lithologies, although settling rates varied in response to slurry densities and site characteristics. Clinker hole slurry columns declined precipitously within 24 hours of emplacement as fractures filled and partially plugged, and then declined at more gradual rates until an equilibrium was established. The final column heights of 8.6 lb/gal and 9.0 lb/gal slurries in clinker were 10% and 30% of original column heights, respectively. In sandstone holes, the 8.6 lb/gal slurry columns also fell to 30% of their original heights within a few days, then gradually recovered to 55% of the emplacement levels. The 9.0 lb/gal slurries performed much better in sandstone holes, retaining 90% of their emplacement levels throughout the monitoring period. Shale hole slurry column heights initially fell to 45% and 60% of the original levels for the 8.6 lb/gal and 9.0 lb/gal slurries, respectively. During the remainder of the monitoring period, the 8.6 lb/gal slurries gradually recovered to 65% of the original emplacement levels, and the 9.0 lb/gal slurries recovered to ≈75% of the original levels.

Standard errors of within-treatment means were less than or equal to 10% of the mean for data generated at the sandstone and shale sites. Data generated at the clinker site exhibited more variability; within-treatment standard errors of up to 39% of the mean were calculated.

Tub Trials

Replicated laboratory treatments were conducted to analyze the comparative ability to withstand pressure applied to the seal provided by each treatment. Fifty-six different replicated treatments are being studied (seven different grouts, each grout subjected to the influence of eight different water chemistries, each trial replicated three times). Figure 4 represents the design of a typical tub trial. The wall of each tub (representing a bore hole) is smooth acrylic, presenting a poor bonding surface. The tubs (height 8 1/2 in.) are tapered from top (7 in.) to bottom (6 in.), thereby providing a minimized resistance to upward pressures. Pressure was applied to the base of each treatment via water or air pressure introduced through a centralized PVC pipe. Eight evenly spaced vents (saw slots) were made from the bottom up, 1/2 in. on each pipe.

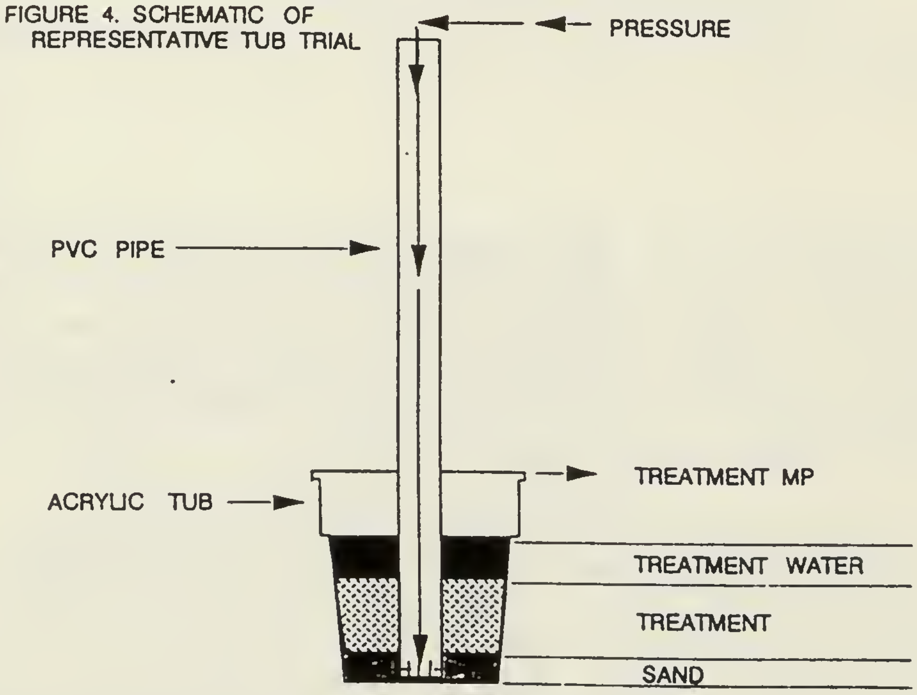

FIGURE 4. Schematic of a tub trial.

The following grouts were tested: 1) cement plus 8% bentonite (mixing ratio to water the same as for straight cement); 2) cement; 3) a 9.0 lb/gal bentonite slurry; 4) a 10.2 lb/gal bentonite slurry; 5) commercial grade bentonite chips; 6) low grade bentonite chip; and 7) a mixture of 25% bentonite (commercial grade) and 75% gravel. The first four treatments were slurries created by mixing water with dry material prior to emplacement in tubs. The remaining three treatments were placed into respective tubs as dry material, and then subjected to saturation by treatment water. Initial thickness of all grout treatments was ≈2.75 in..

Failure of a treatment to withstand applied pressure was noted when a treatment was observed to leak (upward migration of air bubbles or water). Some treatments were lifted slightly by the water pressure in the tub prior to leaking. Failure was not considered to be complete prior to detection of leaking.

Results of pressure trials on the above described grouts blended with Billings city water, are reported here.

Generalized results of the trials are shown as Figure 5. Results depict the comparative initial pressures withstood by each treatment (i.e. treatments subjected to pressure a single time).

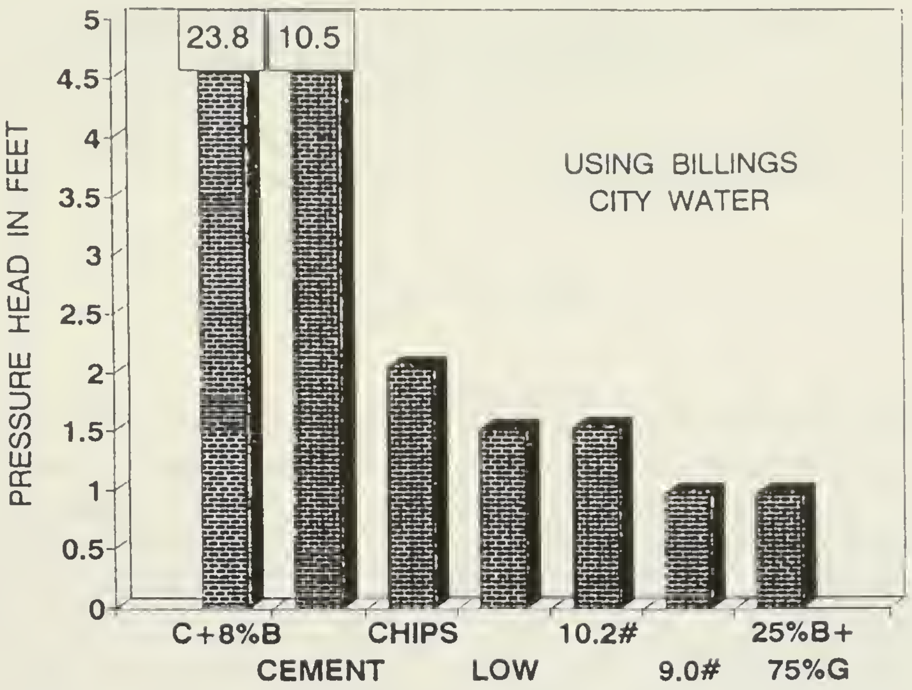

FIGURE 5. Pressure withheld by treatments that were mixed using Billings city water.

Cement plus 8% bentonite provided the most effective pressure seal of any grout material tested. The best of the non-cement treatments, commercial grade bentonite chips, withstood less than 1/10th of the pressure held by cement/bentonite grouts. The seal provided by the cement plus 8% bentonite treatment withstood about twice the pressure (23.8 ft of head) as the neat cement treatment (10.5 ft of head). Commercial grade bentonite chips leaked at 2 ft of head. Both the low grade bentonite chips and the 10.2 lb/gal slurry leaked at 1.5 ft of head. The gravel/bentonite grout and the 9.0 lb/gal slurry each leaked at 1.0 ft of head.

Significance of the tub trial results was tested by one-way ANOVA, and all leakage pressures reported above were different at a significance level of $\alpha=.01$.

CONCLUSIONS

Shallow Hole Trials

The high solids treatments (dry commercial bentonite chip, pit run/low grade bentonite, and the dry gravel/bentonite mix) provided 100+ percent fill in all shallow hole trials. The slurry treatment fills ranged from about 10% for the 8.6

lb/gal slurry at the clinker site, to about 90% for the 9.0 lb/gal slurry at the sandstone site.

Some rise in slurry treatment elevation occurred at the sandstone and shale sites. This rise could be due to a swelling of the natural substrate. This swelling would decrease the diameter of the holes, thereby pressuring the slurries to a higher elevation in the hole. Precipitaion could cause wetting fronts to pass through the shallow bedrock units, allowing extra water to be absorbed and increasing the volume, and therefore the height, of the treatments in the holes. The rise in slurry measuring points could also result from partial sloughing of bore hole walls.

Tub Trials

The cement treatments adhered to the acrylic walls of the tubs and withstood substantial hydrostatic pressure before failing catastrophically. Bentonite proved to be flexible, and in some cases actually moved and still withstood additional hydrostatic pressure before failing. In general, the higher the grade of material, and the higher the solids content of the bentonite treatment, the more pressure the seal could withstand before failing. The lower bentonite content (9.0 lb/gal slurry and the bentonite/gravel mixture) easily allowed flow paths to open up, and water crossed the seal at low hydrostatic pressures.

While the ability of a grout to stand in a bore hole may be desirable in and of itself (stability, prevention of subsidence, etc.), this capability alone does not always provide for all of the intended purposes of grouting. If there is a need to prevent such occurrences as interaquifer mixing, or the escape of gas or flowing waters, the ability of a grout to withstand pressure is an important consideration.

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

A BIOENGINEERING TECHNIQUE FOR STREAM CHANNEL RECONSTRUCTION

Dale Miller and Jim Lovell[1]

ABSTRACT

Cleanup of a small New Jersey stream for heavy metals contamination (mostly lead and mercury) resulted in drastic disturbance to the stream and its associated riparian zone and floodplain. Restoration of the area required designing and reconstructing the stream in 3-dimensional space to a pre-disturbance configuration. Digital terrain models were used to generate a longitudinal profile and cross-sections of the channel at approximately 30 foot intervals. Surface and subsurface substrates were sampled to characterize particle distributions in the streambed. These data were entered into a step-backwater analysis (HEC-2) to evaluate tractive force on the stream bed and banks during flood events. The results revealed that slope through the reach was moderate, about 0.5%, with substrate ranging from silt to gravel. Base flow was less than 1 cubic foot per second (cfs) but the 100-year event was nearly 770 cfs for a channel averaging only 12 feet wide.

Using these data, designs were developed for a new streambed and banks. Rounded gravels were specified for the streambed in a range of sizes from 3/4 to 6 inches in diameter to ensure a non-mobile bed during extreme discharges. Clean topsoil and subsoil were hauled in to bring the floodplain back to near-original grade. Designs for stream bank reconstruction relied on bioengineering principles to develop vegetated banks, reinforced with a dual layer of biodegradable erosion fabric for short-term (5 to 6 years) structural strength. Dimensions of the fabric-reinforced streambanks were determined by stability analysis. Long-term channel stability was provided by extensive revegetation of the banks with grasses, forbs, shrubs, and trees.

This system is currently in its second year of recovery and is functioning well. The bioengineering techniques used to restore the stream are be applicable to other drastically disturbed sites.

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[1] Inter-Fluve, Inc., 25 North Willson Suite 5, Bozeman, MT 59715 Ph: (406) 586-6926.

INTRODUCTION

A small, third-order tributary to the Wanaque River, New Jersey, was treated for heavy metals contamination (mostly lead and mercury). Cleanup plans called for complete removal of contaminated floodplain soils to depths exceeding five feet below original ground surface. As a consequence, the stream and its associated riparian zone and floodplain were drastically disturbed. Restoration of the floodplain required backfilling the excavated areas with clean fill and reconstructing the channel and banks of the stream.

The stream flowed through an undeveloped area and an older residential neighborhood with buildings and landscaping immediately adjacent to the stream. Much of the stream corridor had been modified by property owners, and subsequently encroached upon by an assortment of retaining walls, fill, houses and secondary buildings. Other owners had altered riparian vegetation from native and adapted species to ornamental plants and lawns.

Engineering designs for channel reconstruction often call for applications involving stone riprap or concrete structures. The primary benefit of such applications is a level of channel stability within an acceptable safety margin. Because urban watersheds generally have a high density of residential or commercial buildings, often within the 100-year floodplain, natural channel changes (natural lateral or vertical migration) must be constrained. As a result, a stream reconstructed in this environment cannot migrate appreciably without potential loss of property or structures.

While hard engineering approaches to stream channel reconstruction offer stability, they generally have fewer environmental benefits than alternative, bioengineered approaches. Consequently, bioengineering techniques that meet or exceed acceptable safety standards were adopted for this project. These techniques relied primarily on organic, biodegradable textile fabrics and revegetation with native and native-adapted plants. The fabrics were used in retaining walls and as slope protection to provide short-term stability (less than 5 years), while trees, shrubs and grasses were planted to provide long-term stability (greater than 5 years).

This bioengineering approach allowed the stream to be reconstructed within acceptable stability guidelines, while constraining lateral and vertical movement of the channel. As vegetation matures, the character and configuration of the reconstructed stream will be very similar to pre-disturbance conditions.

HYDROLOGIC AND HYDRAULIC ANALYSIS

In order to reconstruct the stream with a channel that would remain stable during floods, it was necessary to complete hydrologic and hydraulic analyses. A Soil Conservation Service technical release (TR-55) provided a method for estimating the hydrology of small (less than 5 square miles), urban watersheds. The method estimated peak discharges at various frequency intervals from rainfall and time of concentration data for the 512 acre drainage basin. In brief, the 2-year event (approximate bank-full) was estimated to be 128 cubic feet per second (cfs) and the 100-year event was a sizeable 771 cfs.

A HEC-2 (step-back water analysis) also was conducted using photogrammetric topographic data and extensive field surveys of bridges. This HEC-2 run was used for characterization of peak discharges. To characterize low and mid-sized discharges, a second HEC-2 was run based on detailed cross-sectional channel data surveyed in the field. These analyses were critical to design as they provided data on water depth and velocity for a range of discharges at different locations. To ensure that reconstruction designs provided adequate stream bed and bank stability, the 100-year discharge of 771 cfs was used as the design event.

TOPOGRAPHIC INFORMATION

Detailed cross-sections of the channel and floodplain were surveyed within the project area. The stream channel was surveyed in relatively high detail in order to allow reconstruction as closely as possible to the measured configuration. As discussed previously, dense development within the stream floodplain necessitated reconstruction of the stream bed and banks to pre-disturbance orientation. Approximately 145 cross-sections were measured at predominant breaks in slope and changes in channel dimensions. Substrate type, riparian vegetation, large trees, building corners and man-made structures were identified and recorded.

A channel longitudinal profile of the stream was developed from the cross-sectional survey data. The results revealed that slope through the reach was moderate, about 0.5%, and the bed profile was relatively featureless. Channel bedforms, pools and riffles, were for the most part poorly defined. The number and depth of pools was limited, and although some riffles existed many were backwatered by man-placed debris. As expected, fine sediment collected in numerous backwater areas. Overall, streambed substrates ranged from silt to gravel. The profile showed that the grade was also significantly affected by roads and bridges. The bridges generally impacted the channel by creating flattened upstream profiles.

Using the topographic information from the channel cross-sections, a digital terrain model (DTM) of the stream channel and floodplain was developed. The model was generated from AutoCAD® and Surpac®, the latter being a mapping and earthwork software package often used in surface mine design and construction. The DTM produced an accurate map of the site and provided cut and fill estimates for materials specifications.

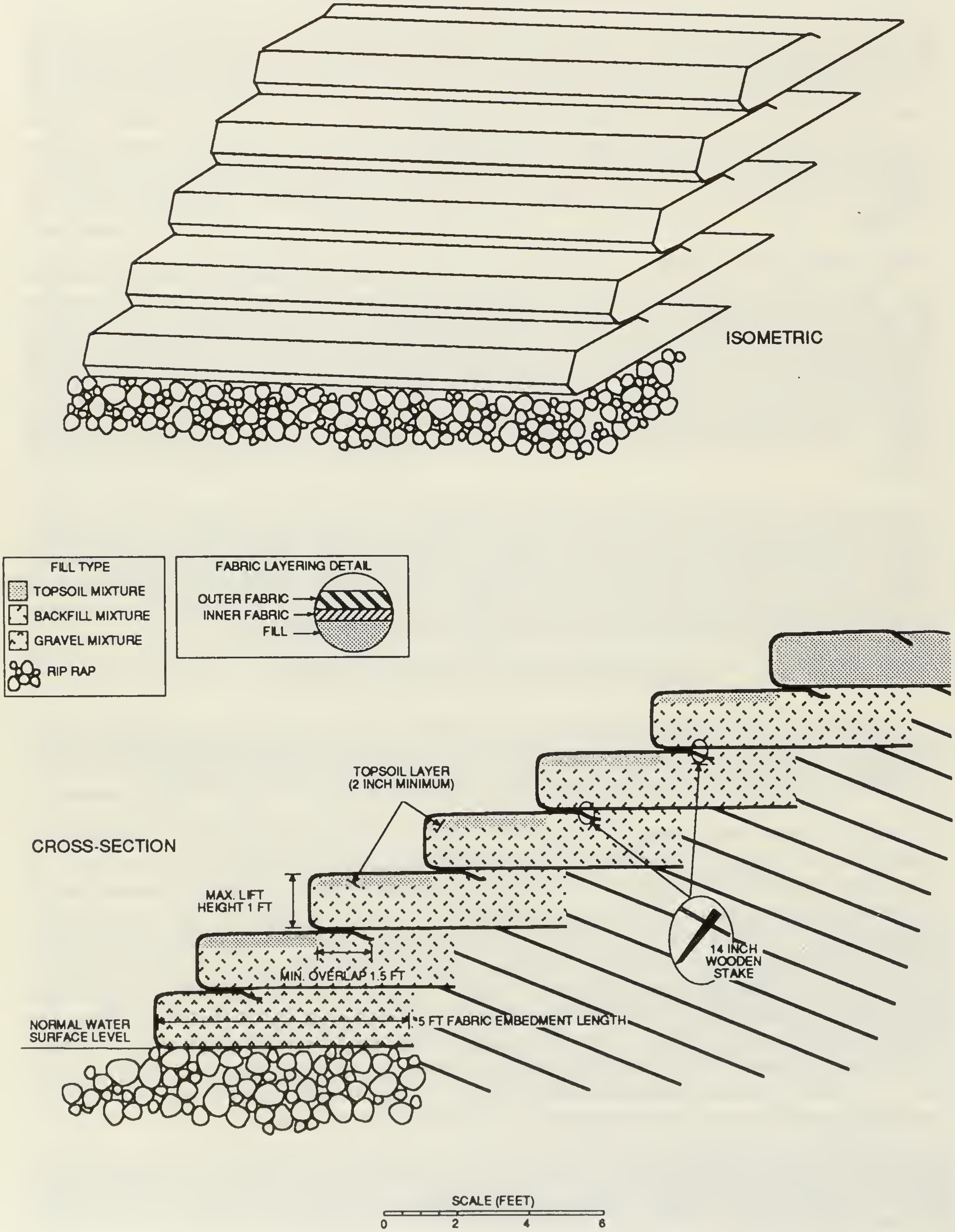
STREAMBANK RECONSTRUCTION

The objective for stream reconstruction was to provide stream bed and bank stability, as well as aesthetic and biologic diversity. This objective was achieved through the use of biotextile materials to reconstruct stream banks and re-establishment of riparian vegetation.

Streambanks were reconstructed in 1 foot high lifts of fill material (Figure 1). Each lift was wrapped in two layers of biotextile fabric to provide stability and preventing piping losses of fill material. Both types of organic, biodegradable fabric were comprised of coconut fabric and will decompose in 5 to 6 years after which planted vegetation will protect the streambanks. The first layer, a coir fabric sold under the trade name DeKoWe® 400 by Belton Industries, Inc., consisted of twisted coconut fibers woven into a strong, durable netting with openings between the warp and weft of about 1/4 inch. This fabric was used on the outer, or exposed layer, of the bank stability treatments. The purpose of this fabric was to provide structural integrity. The second fabric consisted of non-woven coconut fibers loosely intertwined and held together with a plastic, non-photodegradable netting on each side. This fabric was sold under the trade name C125 by North American Green®, and was used as an interior layer to prevent piping of fine clays, silts and sands.

Streambank reconstruction began with surveying to determine the location and slope of the banks. Construction then proceeded from the bottom of the bank to the top in an upstream to downstream direction. First, the fabric layers for the bottom lift were rolled out along the

Figure 1. Isometric veiw and cross-section of reconstructed streambank using biodegradable erosion fabric for stability and short-term bank protection.



long axis of the stream and moved into place according to the survey. Fill was then placed on half of the fabric to a depth of 1 foot and compacted to 90%. The loose half of the fabric was folded back over the fill, keyed into a shallow trench, and fastened in place with 14 inch wooden stakes on 3 foot centers. This procedure was then repeated for the second lift and so on until the top lift was installed.

Stability calculations were undertaken to determine appropriate dimensions of the lifts using calculations for retaining walls at vertical and 45° slopes. Two criteria were considered in order to determine the embedment length of fabric and the vertical spacing of lifts. Near the surface, the embedment of the fabric had to be sufficient to prevent pullout. In deeper lifts, pullout became less critical, but the embedment length had to be sufficient to prevent the retaining wall from sliding. The combination of these two criteria tended to cause the embedment length to remain relatively constant from top to bottom

The embedment length of fabric and the vertical spacing of lifts were calculated using minimum properties of the proposed soil backfill and fabric while assuming a factor of safety of 2. The vertical reinforcement spacing of the lifts was specified to be about 1 foot. Similarly, a minimum embedment length was determined to be 3 feet. However, sliding analysis indicated that for vertical walls higher than about 4 feet an embedment length of 5 feet was necessary. For ease of construction, the following dimensions were used for all fabric retaining walls: vertical spacing = 1 foot; embedment length = 5 feet (Figure 1).

The streambanks were configured so that their slopes and fill materials replicated the pre-disturbed channel configuration. Consequently, the banks may begin at 0:1 slope for 2 lifts, then 2:1 slope for 2 lifts, then 3:1 slope for the last lift. The type of fill used in each lift depended upon the elevation of the lift relative to normal water surface (Figure 1). The bottom lift consisted of washed, rounded stream gravel to provide resistance to scour. The next series of lifts used clean a subsoil backfill with a minimum 2 inch layer of topsoil spread over the top of each lift. The top lift was backfilled entirely with topsoil.

After installing the streambanks, the streambed was reconstructed with washed, rounded, stream gravel ranging from 3/4 to 6 inches in diameter. This material was placed over a 4 inch layer of rock crusher dust which provided a stable base and reintroduced fines to the stream. Without the filter material, water would have more of a tendency to flow between the gravels rather than above the streambed. The streambed was configured to recreate pool, riffle, and bar features at a scale appropriate to the size of the stream being reconstructed.

REVEGETATION

The key to long-term bank stabilization using the techniques described in this paper was the establishment of mature riparian vegetation. Grass, forb, shrub and tree species were specified based on local climatic conditions, proximity to stream channels, and ability to produce dense root systems at maturity. All grass and forb stock was planted by seed and incorporated into the bank stability treatments during construction. Shrub cuttings were placed between stream bank lifts as retaining walls were being constructed. Shrub and tree bare root stock were planted in fabric retaining walls after the walls were constructed. In this way, the integrity of the fabric comprising the retaining walls was not compromised by the planting method.

Because the outsides of bends tend to receive higher erosive energy during flood events, planting density was doubled on the outside of bends with an angle of more than 45°. This increased planting density was extended about 30 feet upstream and downstream of the bend apex.

All exposed fabric was seeded with grasses and forbs by broadcasting prior to placement of final fabric wraps or lifts. Grasses were planted at a combined rate of 20 pounds per acre and forbs were planted at a combined rate of 10 pounds per acre. The percent composition of grasses and forbs is listed in Table 1.

Along the lower streambanks, cuttings of shrubs were planted between fabric lifts at intervals of approximately 0.5 feet. The cuttings were 3 to 5 feet long and 1/2 to 1 inch in diameter and were placed with the butt ends extending 6 to 12 inches between the fabric lifts. The growing direction of each stem was oriented at 45° downstream to prevent debris from accumulating on the plantings during high flows.

Trees and shrubs were also planted as bare root stock on the streambanks and the floodplain. The percent composition for tree and shrub species is listed in Table 1. Along the lower streambanks, trees and shrubs were planted in approximately 50 feet long patches of the same species. On the upper streambanks and on the floodplain, woody vegetation was spaced randomly. Table 1 lists the type of stock and the planting density for each stream area.

SUMMARY

A small stream was disturbed during cleanup of heavy metals. Restoration efforts utilized bioengineering techniques to reconstruct the streambed, banks, and floodplain. These techniques relied on the use of biotextile fabrics to provide short-term stability and erosion protection. Long-term protection was provided by re-establishing riparian vegetation.

Reconstruction of the first 1,000 feet of stream channel and floodplain was completed in March of 1992. After the first six months of growing season, cuttings and rooted stock shrubs and trees were well established. Willow and dogwood cuttings had new stems averaging 2 to 3 feet in length. Grass and forb growth was vigorous, approaching complete coverage. While root growth and ground cover is not expected to reach ideal density for 3 to 4 years after planting, the initial response suggests that stabilization goals were met for the stream reconstruction project. The bioengineering techniques developed for this project are currently being applied to other drastically disturbed stream and embankment sites.

Table 1. Plant species recommended for revegetation in the stream riparian area and floodplain. Also listed are the type of stock and the typical planting density for each species.

LATIN NAME	COMMON NAME	STOCK	PLANTING DENSITY	
Lower banks§				
Salix nigra,	Black willow	bare root cuttings	1 ft centers every 0.5 lineal foot	60%
Cornus amomum,	Silky dogwood	bare root cuttings	1 ft centers every 0.5 lineal foot	20%
Cornus stolonifera,	Red osier dogwood	bare root cuttings	1 ft centers every 0.5 lineal foot	20%
Upper Banks§				
Betula nigra,	River birch	bare root	4 ft centers	50%
Quercus palustris,	Pin oak	bare root	4 ft centers	50%
Cornus amomum,	Silky dogwood	bare root	2 ft centers	50%
Cornus stolonifera,	Red osier dogwood	bare root	2 ft centers	50%
Floodplain				
Acer rubrum,	Red maple	bare root	6 ft centers	25%
Platanus occidentalis,	Sycamore	bare root	6 ft centers	25%
Fraxinus pennsylvanica,	Red ash	bare root	6 ft centers	25%
Robinia psuedo-acacia,	Black locust	bare root	6 ft centers	25%
Ligustrum vulgare,	Privet	seed	3 ft centers	50%
Rhus glabra,	Smooth sumac	seed	3 ft centers	50%
*Include In All Zones				
Impatiens capensis,	Jewelweed	seed		16%
Chelone glabra,	Turtle head	seed	*total for all forbs 0.23 lbs per 1,000 sq ft*	16%
Eupatorium maculatum,	Spotted joe-pyeweed	seed		16%
Boehmeria cylindrica,	False nettle	seed		16%
Hemerocallis fulva,	Day lilly	seed		16%
Lotus corniculatus,	Birdsfoot trefoil	seed		16%
Festuca arundinacea,	Tall fescue	seed	*total for all grasses 0.46 lbs per 1,000 sq ft*	25%
Poa pratensis,	Kentucky bluegrass	seed		25%
Lolium perenne,	Perenial ryegrass	seed		25%
Agrostis alba,	Red top	seed		25%

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§ Planting density should be doubled on the outside of bends, 20 feet upstream and 30 feet downstream of bend apex.

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

RAISING THE WATER TABLE TO CREATE A WETLAND

Steve McIntosh

ABSTRACT

Restoration of a wetland which was initially disturbed at the turn-of-the-century will be undertaken to compensate for the temporary loss of wetlands associated with future mining operations. The proposed mitigation area exhibits xeric conditions as a result of past dredging and gravel mining operations which had the effect of lowering the flood plain surface and wasting the fine surface sediments. Together these mining activities resulted in complete alteration of the surface and subsurface water systems. Water which had inundated the soil profile, now flows on the basalt bedrock, surfacing only for a relatively short time during high flow periods.

The conventional approach to wetland restoration in this type of area is to remove surface material in an attempt to bring the surface nearer to the underground water. These operations would require the removal and redistribution of large quantities of gravel, cobbles and boulders and the attendant destruction of existing wetland areas. Rather than lowering the surface elevation to the water level, the mitigation plan conceptualized for this project involves forcing the water to the surface by creating subsurface ponds behind four impermeable barriers. The barriers will extend vertically from the bedrock to the surface. Surface berms will be constructed along the length of each groundwater barrier to enhance sediment deposition and encourage the development of wetland vegetation. This is the least expensive and least invasive method of creating a wetland which will remain saturated to the requisite degree and for the amount of time required for classification as a wetland as defined by the 1989 Federal Manual for Identifying and Delineating Jurisdictional Wetlands.

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[1] Environmental Engineer, NERCO DeLamar Mine, Jordan Valley, OR.

INTRODUCTION

Compensation for future mining disturbances is addressed by a wetland restoration plan which focuses on elevating the groundwater level in a turn-of-the-century dredge mine area. The groundwater will be raised by installation of a series of subsurface flow barriers. Surface berms will be constructed above each barrier to slow overland flows associated with major flow events and encourage sediment deposition. Staged weirs are to be placed in-line with each groundwater barrier to control the escape of captured water. Placement of the barrier/berm/weir systems will be based on the amount of water required for creation of a wetland.

This wetland construction approach is somewhat unconventional. Typically either the surface material is lowered to bring it in closer contact with the groundwater or the surface water area is increased by construction of a reservoir. Neither of these options are appropriate in this setting.

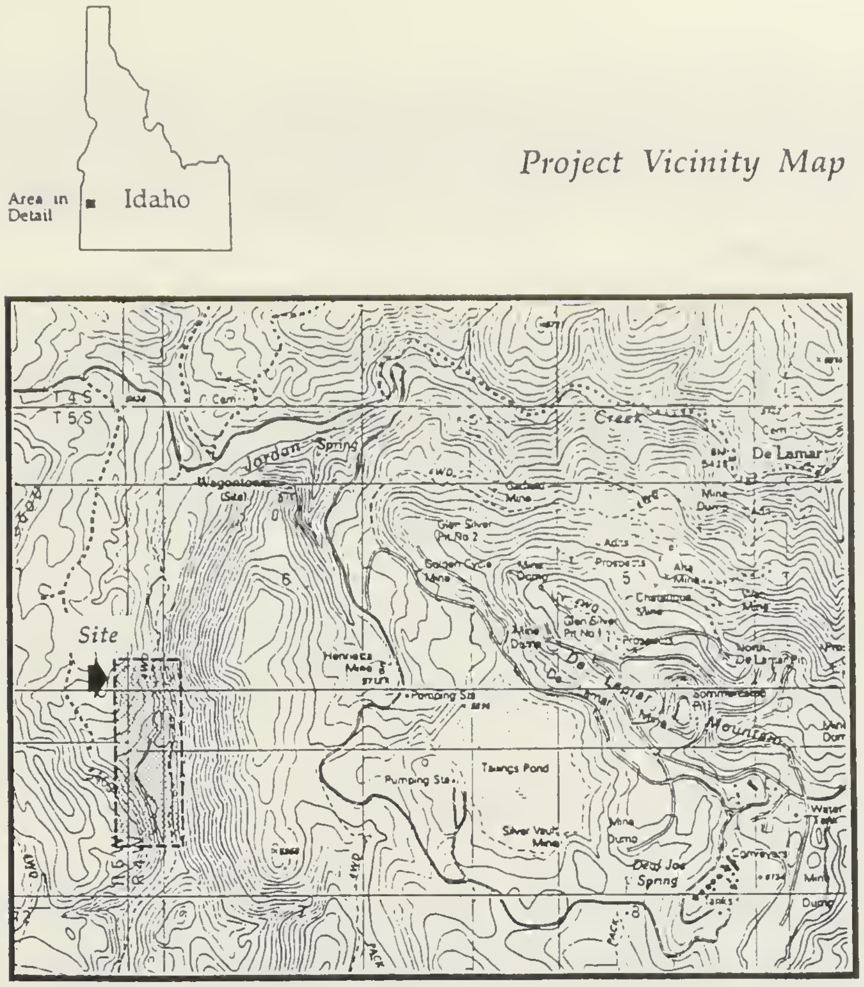
DESCRIPTION OF THE MITIGATION SITE

Location

The proposed mitigation site is located in the Owyhee Mountains in southwestern Idaho between Jordan Valley, Oregon and Silver City, Idaho (Figure 1). The north end of the project area is about 3,000 feet downstream of the DeLamar Road within the alluvial floor of a north-south trending valley. Valley relief averages 800 feet from the valley floor to the nearby summits. The valley floor of the wetland mitigation study area ranges from 200 to 500 feet wide. Jordan Creek flows along the west side of the valley. Though classified as a perennial stream by the USGS, Jordan Creek exhibits intermittent flow through the year as a result of past mining activities.

Surface Materials

Prelaw dredge mining operations inverted and mixed the entire soil profile. As a result the area is characterized by an undifferentiated mixture of sands, gravels cobbles and boulders. These materials rest on an impervious layer of basalt. The unconsolidated surface material ranges in depth between 5 and 10 feet and has extremely high conductivity and permeabilities. No barriers to water flow exist between Jordan Creek and the valley. A direct relationship therefore exists between the subsurface water in the valley and Jordan Creek. Two key properties exist in the project area because of these characteristics. First, the water holding capacity of these soils is extremely low. Secondly, the high conductivity of the sediments suggests a direct relationship between the surface water (Jordan Creek) and the ground water.



Project Vicinity Map

Source: USGS 7 1/2 minute quadrangle, Delamar 1990 (Provisional Edition)

Figure 1

Proposed Wetland Mitigation Area for the Stone Cabin Mine

Figure 1.

Jordan Creek Aquatic System

Jordan Creek is a perennial stream that has been drastically altered by past mining operations. Placer mining operations were hindered by the natural sinuous flow pattern of Jordan Creek within the valley. To increase the size of the dry working area, the dredge miners confined Jordan Creek to the west side of the valley by constructing a levee 5 to 14 feet in height. The channelized waters cut to the basalt layer and caused a dramatic drop in the surface water elevation.

Wildlife

Because of the direct relationship between the surface and groundwater, the drop in the surface water level has resulted in a corresponding drop in the groundwater level. Subsequent gravel mining operations removed a significant amount of valley fill, lowering the surface elevation in the project area. In combination these disturbances caused the development of a surface water system that is dry much of the year. The majority of the water which passes through the valley flows below the surface along the basalt contact.

During dry periods, small pools of stagnant water are all that remain in the channel. These pools are characterized by high water temperatures and low oxygen levels. Aquatic biota are seriously impacted by these effects. Of particular concern is the redband trout.

Redband trout (*Salmo gairdnerii*) are a variety of rainbow trout that have evolved for survival in the harsh semiarid conditions typical of eastern Oregon and southern Idaho (Coffin 1980, Gold 1980). Redband trout will die during low flow periods due to the elevated water temperatures and limited oxygen levels which characterize the pools of water which remain in Jordan Creek during drought periods. Restoration of year-round flow in the area would result in a marked improvement in the health of the aquatic biota. Elevation of the groundwater level is the preferred method of restoring the surface water regime in the project area.

Vegetation

Upland, semiarid rangeland grass species dominate the project site. Kentucky bluegrass, intermediate wheatgrass, mountain timothy, junegrass, and tufted hairgrass predominate. Willows are relic wetland species that can be found in some areas. Common forbs include mullen (*Verbascum thaspus*), buckwheat (*Eriogonum* spp.), and dock (*Rumex acetosa*).

Small pockets of wetland areas dot the project site. These are largely composed of pedestals and low spots left relatively undisturbed by past mining activities. Species in these areas include: willow (*Salix exigua, S. commutata, S. lasiandra*), red-osier dogwood (*Cornus stolonifera*), (the latter genus are facultative or obligate wetland species), Rocky Mountain maple (*Acer glabrum*), streambank wheatgrass (*Agropyron trachycaulum*), raspberry (*Rubus stragosus*) and meadow rue (*Thalictrum venulosum*).

GENERAL WETLAND CONSTRUCTION APPROACHES

Two general approaches are available for creation of wetland conditions in the project area; (1) remove enough surface material to bring the surface closer to the water or (2) force the water closer to the present surface. In the first case, the wetland design is based on detailed analyses of the characteristics of adjacent waterways with the intention of copying the appropriate stream channel characteristics within the construction area. This involves an extensive amount of soils and overburden movement. Furthermore the longevity of these contrived waterways is normally very limited, especially in headwater areas where new channel meanders develop quickly. This is unacceptable in terms of the Corps of Engineers wetland construction requirement for in-perpetuity (FICA 1989).

The second general approach to construction typically involves installation of a classic, surface impoundment. This option is undesirable for several key reasons. The foremost consideration is the impact on redband trout. It is critical that passage for the trout be relatively obstacle-free.

A two (2) foot height limit has been set in Jordan Creek for obstacles within the stream channel. This figure is based on the "jumping" capabilities of the trout (Nemenyi 1941). Ladders or bypass structures must be installed to address fish passage on a dam. These are high-maintenance features that conflict with the requirement for a self-maintaining, permanent wetland construction program (EPA 1990).

Another major reason a surface reservoir is undesirable is the fact that only saturated zones and relatively shallow water areas are counted as wetlands. Water depths greater than six (6) feet do not qualify as a wetland. Given this restriction, a very limited amount of acreage would be counted as a wetland in the project area if a classic surface dam were constructed. A final shortcoming is the fact that wetland construction projects in the western states which have been based on water impoundments have typically received relatively few "credits" according to the rating system employed by the Corps of Engineers.

The goal of any wetland mitigation plan is to receive a high "trade-off" value for the project. This number represents the ratio between the mitigation acreage that must be constructed and the acreage affected. The more valuable the wetland project is considered the more "credits" are allocated. This is quantified in a ratio which is commonly referred to as the trade-off value. The more equal the ratio the more acceptable the project is deemed by the review agencies.

A great many of the wetland mitigation projects in the region have received a 4:1 trade-off value. This means that four (4) acres of wetlands must be created in order to be granted the ability to disturb one (1) acre. Therefore this value is very important since it implies a direct monetary cost. A project with a more equal trade-off value is generally less expensive to install since fewer land issues and construction costs are involved.

The ideal wetland construction project would result in: (1) maximization of the wetlands which are created, (2) minimization of the amount of existing wetland acreage which is disturbed by construction of the wetland, (3) minimization of the construction costs (4) and maximization of the trade-off value received for the project. These factors dictate that the most effective wetland construction approach in the project area would involve an alternative to the traditional wetland construction methods. The most appropriate method of creating a wetland in the project area will involve raising the groundwater level to the required height by forcing the groundwater closer to the surface.

PROPOSED MITIGATION PLAN

Construction Format

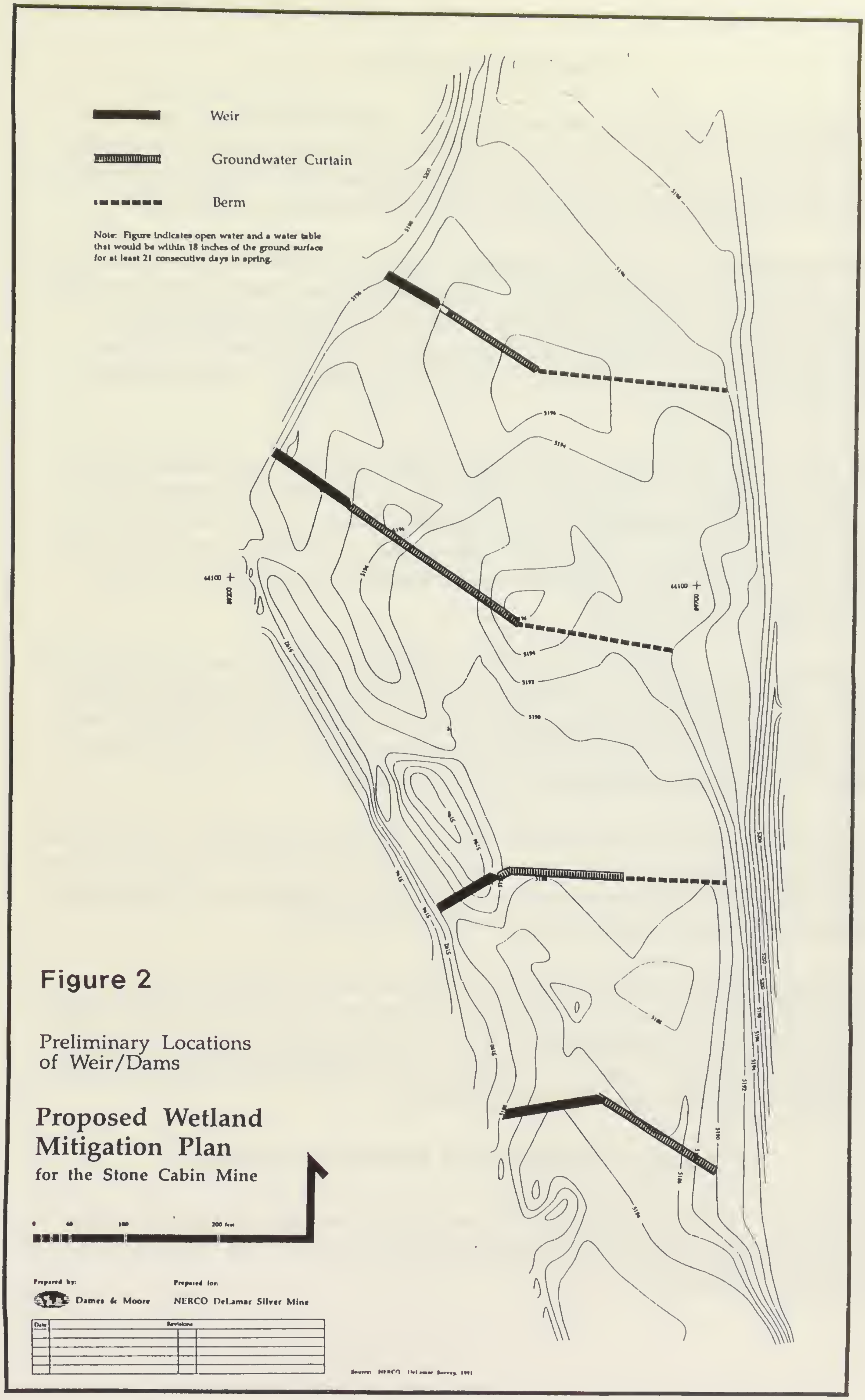
A series of groundwater flow control systems will be installed at select locations in the project area. Each system is composed of: (1) a groundwater barrier that extends from the impermeable basalt to the surface (2) a 2 foot high berm above the barrier and (3) a staged weir which is located on one end of the groundwater barrier in Jordan Creek (Figure 2). Both ends of the barriers are keyed into the steep valley walls to prevent water from bypassing the system.

Each weir and groundwater barrier system has structural components which are analogous to a surface dam with a spillway. The embankment of the surface dam performs the same structural function as the subsurface barrier. Both are designed to obstruct water flow. The spillway on the surface dam serves the same purpose as the staged weir. They both provide for a measured release of captured water.

The weirs control the volume of water which is held behind each dam. The weir is constructed with stages to control the length of time the water is held behind the barrier. The largest stage allows passage of the 100 year flood event. The second stage is wide enough to permit the exit of the 50 year event and the smallest V-shaped weir is sized for the 2 year flows. This system will provide for a measured release of captured waters.

The final design consideration is the distance between each weir and barrier. Two criteria were used to calculate this distance, the Corps of Engineers definition of a wetland and the high water mark for each weir/barrier/berm system. According to the 1989 Corps of Engineers Wetland Delineation manual, the surface of a wetland area must be inundated or saturated to within about 18 inches of the surface for at least 7 days in order to qualify as a wetland (these figures are derived from calculations which are based on the length of growing season, soil type etc. within the project area).

The height of the underground reservoir and the length of time the ponded water remains at a given height must be determined. Essentially the subsurface dam is a mirror image of a surface dam. For a surface dam the critical measure is depth of water. This is equal to the distance from the water surface to the pond bottom. For a subsurface dam the critical measure is the distance from the high water mark to the soil surface. At the point upstream where this distance is equal to 18 inches, another subsurface dam must be installed. Installation of a dam further upstream than this point would result in a non-wetland area where the subsurface water level is further below the surface than allowed by definition.



Note: Figure indicates open water and a water table that would be within 18 inches of the ground surface for at least 21 consecutive days in spring.

Figure 2

Preliminary Locations
of Weir/Dams

Proposed Wetland
Mitigation Plan
for the Stone Cabin Mine

Prepared by:
Dames & Moore

Prepared for:
NERCO DeLamar Silver Mine

Date	Revisions		

Source: NERCO DeLamar Survey, 1991

Feasibility of the Plan

The feasibility of the proposed wetland project is demonstrated by construction projects which took place in the same dredge workings about 6,000 feet upstream from the project area (Study Area). The unintentional effect of these projects was the creation of a wetland. The evolution of this wetland area is instructive.

A raised and compacted roadway was constructed across the dredged area in early 1986 to provide access to a foot bridge. The flow of groundwater down the valley is obstructed by the roadway. However, there was no apparent indication of wetland development following road construction. It was not until a weir was installed near the roadway in Jordan Creek that indications of wetland development became noticeable.

Shortly after road construction (late 1986), a steel, rectangular-notched weir was sealed to the subsurface basalt about 30 feet upstream from the roadway. The weir completed the groundwater control system. The elevated roadway serves as the berm and because it is compacted to the basalt, the roadway also acts as the groundwater barrier. The rectangular weir is identical in function to the weir which is suggested for this plan.

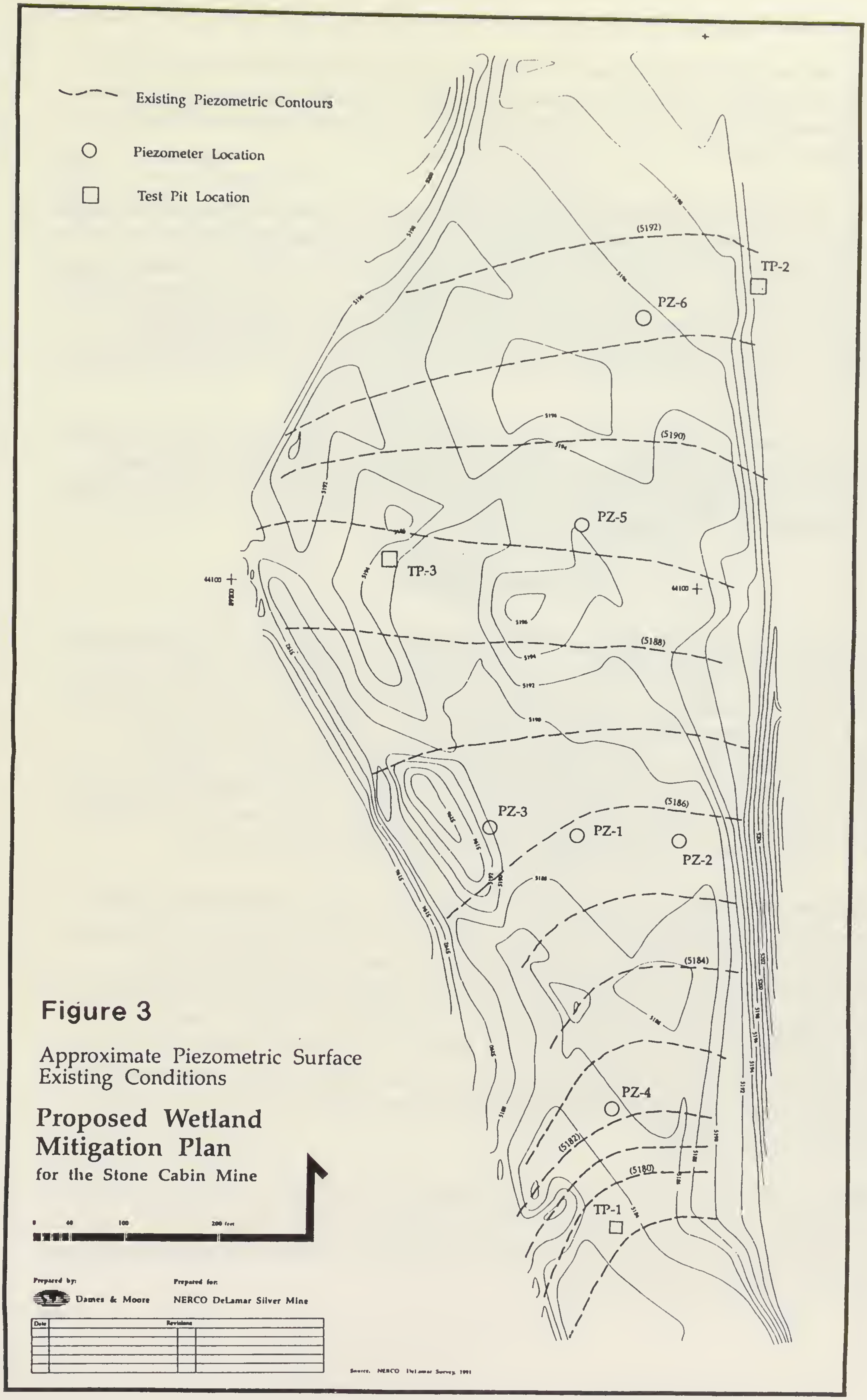
Within one (1) year of weir installation, the improved water status in the area was obvious. The weir blocked the thalweg, or low spot, in the groundwater barrier system. The low spot in the Study Area, as in the project area, is Jordan Creek. The low spot in the weir/berm/groundwater barrier system in the Study Area at this time is the gap between the weir and the roadway.

The gap between the roadway and the weir is particularly instructive. Water exits the wetland area through this gap even during drought periods. The water enters Jordan Creek below the ground surface, appearing as spring water. Only during high flow events does overland flow funnel through this gap.

Repeated observations indicate that the subsurface flow through the gap is nearly continuous and occurs through the majority of the year. The effectiveness of the subsurface dam/berm and weir combination is obvious. The only deviation from this incidental wetland re-construction effort and the proposed wetland construction project will be the materials of construction.

Resultant Change to the Hydrologic Regime

December water level elevations and stream surface water elevations along Jordan Creek were used to prepare the piezometric surface map of the proposed wetland mitigation site (Figure 3).



As indicated on this map the water within the project area tends to follow the topographic contours. The water descends down the valley and perpendicular to the valley walls (NERCO 1991). These characteristics are indicative of a canal, not a wetland area.

Figure 4 illustrates the conceptual piezometric surface in the valley resulting from installment of the barrier/berm/weir system. Also conceptualized are the wetlands which will develop with minimal site grading. This soil movement operation simply entails levelling the high spots within the project site. This grading operation involves low volumes of materials. These areas do not support wetland vegetation. Therefore the existing wetland areas are not disturbed by these soil movement operations.

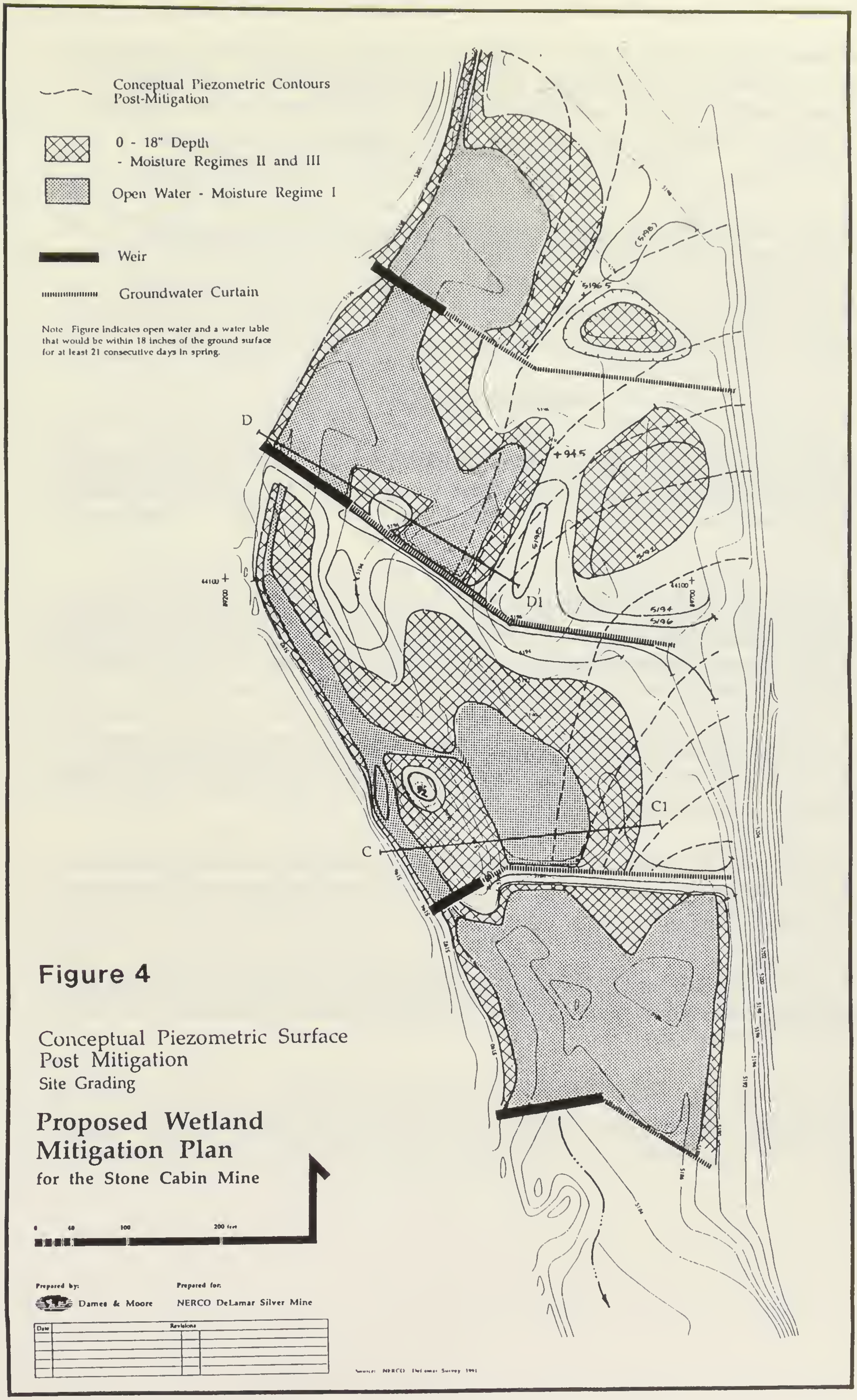
Potential wetland areas created by the increased water table have been estimated by evaluating the groundwater elevation and topography. Based on this evaluation, with minimal grading there would be almost ten (10) acres of saturated areas within 18 inches of the surface (Figure 4).

CONCLUSION

Two wetland construction options are typically used to increase the water level in dredge mine workings. The first is construction of a surface dam. The second option involves re-building the stream course using engineered values for channel features such as flood plains, low-flow courses etc.. After a review of these alternative construction options it was determined that neither of these approaches is suitable in the mitigation area. The reconstruction approach which is ideally suited to restoring the original hydraulic system in the most cost effective manner involves raising the groundwater level within the project area.

The proposed method of wetland restoration represents a modification of the classic surface dam structure. Each feature in the surface dam has a counterpart in this construction plan. The embankment outslope on the classic dam is replaced by an impermeable barrier. The dam spillway will be replaced by a series of staged weirs. The major difference between the surface dam and the groundwater barrier is the placement. In order to capture the subsurface water flow the dam feature must be placed underground. Rather than extending from the ground surface upward, this plan calls for construction from the subsurface basalt to the land surface.

A similar groundwater control system upstream from the project area resulted in the incidental creation of a wetland area providing proof of the feasibility of this plan. The method of wetland construction which was used in this area served as the basis for this construction plan.


Conceptual Piezometric Contours Post-Mitigation
0 - 18" Depth - Moisture Regimes II and III
Open Water - Moisture Regime I
Weir
Groundwater Curtain
D
D1
C
C1
5196.5
(5198)
+94.5
5190
5192
5194
5196
4100
9700
0
40
100
200 feet

Note Figure indicates open water and a water table that would be within 18 inches of the ground surface for at least 21 consecutive days in spring.

Figure 4

Conceptual Piezometric Surface
Post Mitigation
Site Grading

Proposed Wetland Mitigation Plan
for the Stone Cabin Mine

Prepared by: Dames & Moore

Prepared for: NERCO DeLamar Silver Mine

Date	Revisions		

Source: NERCO DeLamar Survey 1991

The increased groundwater levels will provide a basis for establishing wetlands in the valley. Appropriate revegetation following construction of the groundwater barrier system will further enhance the wetland characteristics. The ultimate effect of this work will be the restoration of the wetland characteristics in a historic wetland.

According to the wetland mitigation appraisal system currently recognized by the Corps of Engineers, restoration of the wetland characteristics in areas which formerly supported wetlands are awarded the highest value. The reviewing agencies have suggested a trade-off value of 1.5:1 for this project, verifying the acceptability of the project and the belief in the feasibility of the plan.

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USFWS. February 20, 1991. Project, 1-4-92-SP-245-1019.3031. Correspondence to: Jay Carlson, Owyhee Area Manager, Boise District Bureau of Land Management, 3948 Development Avenue, Boise, Idaho 83708, RE: Stone Cabin Mine

Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium 1992

Evaluation of Organic Substrates for Use In Wetlands Constructed to Treat Acid Mine Drainage[1]

Edward Spotts[2], T.S. Mitchell[2], C.T. Hoschouer[2], and W.M. Schafer[2]

ABSTRACT

A two-phase study was performed to evaluate several commercially available organic substrates for their ability to support the growth of sulfate reducing bacteria (SRB) and improve acid mine drainage (AMD) water quality. In both phases a saturated-flow column technique was employed in which natural AMD was introduced into the columns continually for 6 months. The substrates were examined for their potential use in wetlands constructed to treat AMD.

Results from Phase I, which utilized composted sewage sludge/wood waste, mushroom compost and peat, indicated that all substrates initially increased effluent pH and decreased metals due primarily to inherent neutralization potential (NP) and sorption mechanisms, respectively. Only the mushroom composts, however, maintained circumneutral pH values and reduced metal concentrations in effluent over the long-term duration of the study (6 months). Alkalinity was derived mostly from inherent NP of the substrates and to a lesser degree sulfate reduction. Metals were predominantly retained in the substrates as sorbed and solid phases including hydroxyoxides and carbonates, with only a minor percentage of metals present as sulfides, indicating the role of SRB's metal removal is minor. All substrates in both study phases were net manganese producers.

Because mushroom compost is not readily available at a competitive cost in Montana, several alternative, commercially available substrates were evaluated in Phase II of the study. Substrates included composted wood waste/sewage sludge, peat, straw, cattle manure, sugar beet waste and dried poultry waste. Results indicated that only the mix containing 15 % volumetric dried poultry waste significantly reduced metal levels and maintained a neutral pH in effluent throughout the duration of the study. The high NP of the dried poultry waste substrate mixture was in part responsible for this performance. No metal fractionation analyses were performed on Phase II substrates. In summary, the results of the study suggest that the use of sulfate-reducing wetlands alone may not be sufficient to treat moderate to severe cases of AMD.

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[1] This work was supported wholly by the Montana Department of State Lands, Abandoned Mine Reclamation Bureau. This support was administered by the Abandoned Mine Reclamation Bureau from funds returned to Montana from the Office of Surface Mining of the U.S. Department of Interior.

[2] - Schafer and Associates, 865 Technology Blvd., Bozeman, MT 59715.

INTRODUCTION

Acid mine drainage (AMD) is present at many abandoned coal and hardrock mines throughout Montana and results when pyrite-rich coal, ore or associated overburden is exposed to oxidizing (weathering) conditions. The result is an acidic effluent characterized by low pH (2 to 4) and elevated levels of TDS, sulfate, aluminum, iron, manganese and possibly other trace metals (British Columbia ARD Task Force 1989). In the past few years, the use of constructed wetlands as a passive means of treating AMD has received considerable attention (Girts and Kleinmann 1986, Girts et al 1987, Hedin et al 1988, Wieder et al 1990, Taddeo and Wieder 1991, Calabrese et al 1991 and others). A critical component of the wetland system is the organic substrate. Most constructed wetlands have been designed as throughflow systems, where AMD flows over a substrate and metals are removed primarily via oxidation and the formation of metal hydroxyoxides. Often there is very little increase in pH. Conversely, wetlands designed to maximize contact of the AMD with the substrate to optimize microbial sulfate reduction have been observed to produce a notable increase in pH in addition to metal removal. To optimize sulfate reduction, a substrate must provide a readily-metabolized source of organic carbon (C) and adequate nitrogen (N) for the sulfate reducing bacteria. Additionally, the substrate should support the growth of cattails or other aquatic plants which will become a long-term source of C and nutrients for the bacteria. Uniform physical structure and moderate to rapid hydraulic conductivity (10^{-4} to 10^{-3} cm/s) are also a necessary attribute in order to prevent short-circuiting of upflow and to allow for maximum contact with the substrate.

Schafer and Associates was retained by the Montana Abandoned Mine Reclamation Bureau (AMR) to perform a screening and evaluation of potential substrates for use in constructed wetlands to treat AMD. A bottom-feed column experimental approach was employed using AMD collected from the combined flows of the French Coulee/Anaconda Mine adit discharges in Belt, Montana. The study was completed in two phases. This article presents the results of the Phase II study.

The Phase I wetland substrate column study (Schafer and Associates 1991) was performed to evaluate the effectiveness of several commercially available substrates, including 2 mixed wood waste/sewage sludges, 2 spent mushroom composts and 2 peats, at treating acid mine drainage (AMD) and supporting microbial sulfate reduction. Column effluent and substrate metal fractionation results indicated that the mushroom composts significantly outperformed the wood waste/sewage sludges and peats at removing metals (especially Fe and Al), increasing alkalinity and pH, and supporting sulfate reduction.

Prior to the initiation of the Phase II study, the cost of several potential mixes of substrate was compared to determine which mixes would be most cost-effective. All materials used in the mixes are produced and available in Montana, with the exception of dried poultry waste (DPW). This information, in conjunction with pertinent substrate physical and chemical characteristics, was used to select 6 separate substrate mixes for use in the Phase II study. In addition to developing a cost-effective substrate for use in a constructed wetland AMD treatment system, specific objectives of the study were to quantify the performance of each organic wetland substrate mixture at improving AMD water quality and to determine the biogeochemical mechanisms responsible for substrate performance.

MATERIALS

Acid Mine Drainage

Acid mine drainage used in the study was collected from the combined discharge of French Coulee and the Anaconda Mine (abandoned coal adits) located in the southwest quadrant of the town of Belt, Montana. Effluent was collected in 5 gallon portable jugs and brought back to the lab for use in the columns. The mean chemistry of influent AMD is summarized in Table 3.1. It should be noted that the chemical composition of the AMD used in the study is considerably lower in metals, acidity, and sulfate than actual samples due to metal precipitation reactions during storage.

Table 3.1. **Mean AMD influent chemistry.**

Al	Fe	Mn	Zn	SO_4	Acidity as $CaCO_3$	pH
---------------------- mg/l ----------------------						SI
84.6	95.6	0.41	2.97	912	523	2.67

Substrates

Six separate substrate components were used to formulate the 6 substrate mixes. These components include wheat straw, cattle manure, Farmers peat, Eko-Compost, dried poultry waste and sugar beet waste. Individual components of the substrate were chosen to provide a readily soluble (manure) as well as long term (peat, straw, wood waste) source of C and N. Sugar beet waste, from Billings, was chosen to provide an immediately available source of C to accelerate microbial establishment.

The 6 substrate mixes are shown in Table 3.2. Mixes were designed to allow a comparison between Eko-Compost, peat and straw as a long term substrate "base"; a comparison between cattle manure and dried poultry waste; the effect of the percentage of manure in substrate; and the effect of the addition of sugar beat waste to the substrate.

Table 3.2. **Percent of individual substrate components in each substrate mix.**

Substrate Mix (Column)	Cattle Manure	Straw	Eko-Compost	Peat	DPW[1]	SBW[2]
(1) Manure/straw	30	70				
(2) Manure/straw/Eko-Compost	30	35	35			
(3) Manure/straw/peat	30	35		35		
(4) Manure/peat	60			40		
(5) Manure/peat/DPW		45		40	15	
(6) Manure/straw/peat/ SBW	30	30		30		10

1 - Dried poultry waste
2 - Sugar beet waste

Column Apparatus

Each substrate mix was placed into a column to a depth of 80 cm. The columns used in the study were constructed of 4 inch ID clear acrylic. At the base of each column an influent port was installed through which AMD was introduced into the column. A similar effluent port was located approximately 10 cm above the top of the substrate at the top of the column. Each port consisted of a hollow plastic nipple threaded into the column sidewall and attached to 1/2 inch tygon tubing. The tubing from the influent port lead to a 20 liter influent carboy from which the AMD was fed into the column. At the top of each carboy was secured a Mariotte device to maintain constant head. Thus, by moving the Mariotte tube up or down, the head was adjusted to obtain the desired flow rate. Ceramic cup depth sampling ports were also installed in each column at 3, 8, 16, 30, 47 and 77 cm above the base of the substrate. However, no data from these ports is reported in this paper.

METHODS

Eight cm of 1 to 3 inch washed gravel was placed into the bottom of each column and a total of 80 cm of each substrate mix was added in two 45 cm lifts and compacted with 4.5 psi between each lift to achieve uniform bulk density and porosity within and between each column. After the substrate was in place, AMD was added to each column and allowed to remain in the columns for 10 days to establish equilibrium conditions prior to the introduction of any additional AMD. Flow was be controlled by adjustments in head through manipulation of the Mariotte device.

Flow rate in all columns was reduced approximately 50 % after 66 days to increase residence time roughly twofold. It is important to note that at the end of the equilibration period, but prior to flow initiation, column 5, which contained the manure/peat/DPW substrate mix, toppled over and was broken. Replacement parts were ordered and a new column apparatus was constructed. This resulted in a delay between the start up of column 5 relative to the other columns. Flow was initiated in column 5 twenty days after the other columns. All data interpretations and graphs take into account this delay in the discussion of elapsed time from the initiation of AMD flow.

Random samples of influent were collected periodically and analyzed for dissolved Al, Mn, Zn and Fe, including periodic analysis of Fe $^{2+}$ and Fe^{3+}, using inductively couple plasma spectrophotometry (ICP). A second unfiltered sample was also collected and analyzed for SO_4, acidity, alkalinity, TDS and periodically for NO_3-nitrogen, total Keldhal nitrogen (TKN) and total organic carbon (TOC). Effluent pH, EC, Eh and temperature were regularly recorded in solution at the top of each column. Effluent samples were collected approximately monthly and submitted for the same analyses as the influent.

Columns 1 through 4 and 6 were inoculated on Day 121 with sulfate reducing bacteria (SRB's) collecting with a syringe from the top 3 ports in column 5, where sulfate reduction was observed to be occurring. Ten ml of the collected solution was then injected into the lower sampling port of columns 1 through 4 and column 6.

Inherent substrate neutralization potential was also determined for each substrate mix and for each individual substrate component prior to use in the column by titrating a 20 gram sample with 1 N Hcl to an end point pH of 2.0. The amount of Hcl necessary to decrease the pH to 2.0 was recorded and from this a neutralization potential in $CaCO_3$ equivalents was calculated.

RESULTS

Iron and Aluminum

Although most substrates exhibited good initial removal of Fe (38 - 99 %), eventually all substrates, with the exception of manure/peat/DPW, showed a general decrease in removal efficiency by Day 60. This decrease in removal efficiency remained relatively unchanged through completion of the study (Figure 1). By Day 60, effluent concentrations of Fe had increased in all substrates except manure/peat/DPW and removal efficiencies decreased to between -43 and 19 %. The manure/straw/peat substrate generally outperformed all other substrates with the exception of manure/peat/DPW and manure/straw (in final sampling round) throughout the study. The addition of SBW had no significant effect on effluent Fe removal efficiency, as evidenced by a comparison with manure/straw/peat effluent results.

Iron removal performance in the manure/peat/DPW mix was similar to it's performance in the removal of Al, with > 95 % removal efficiency. Again, the strong performance of the DPW additive substrate was likely the result of a combination of sorption mechanisms and circumneutral pH values. Effluent SO_4 results, which show a considerable reduction versus mean influent levels relative to the other columns, and the maintenance of neutral pH values beyond the time frame expected if only inherent substrate NP were responsible for pH control, suggest that sulfate reduction was occurring in column 5 and that removal of Fe was due in part to the formation of FeS and or FeS_2, along with Fe hydroxyoxides. The observation of several "black spot" regions within column 5 was also indicative of SRB activity.

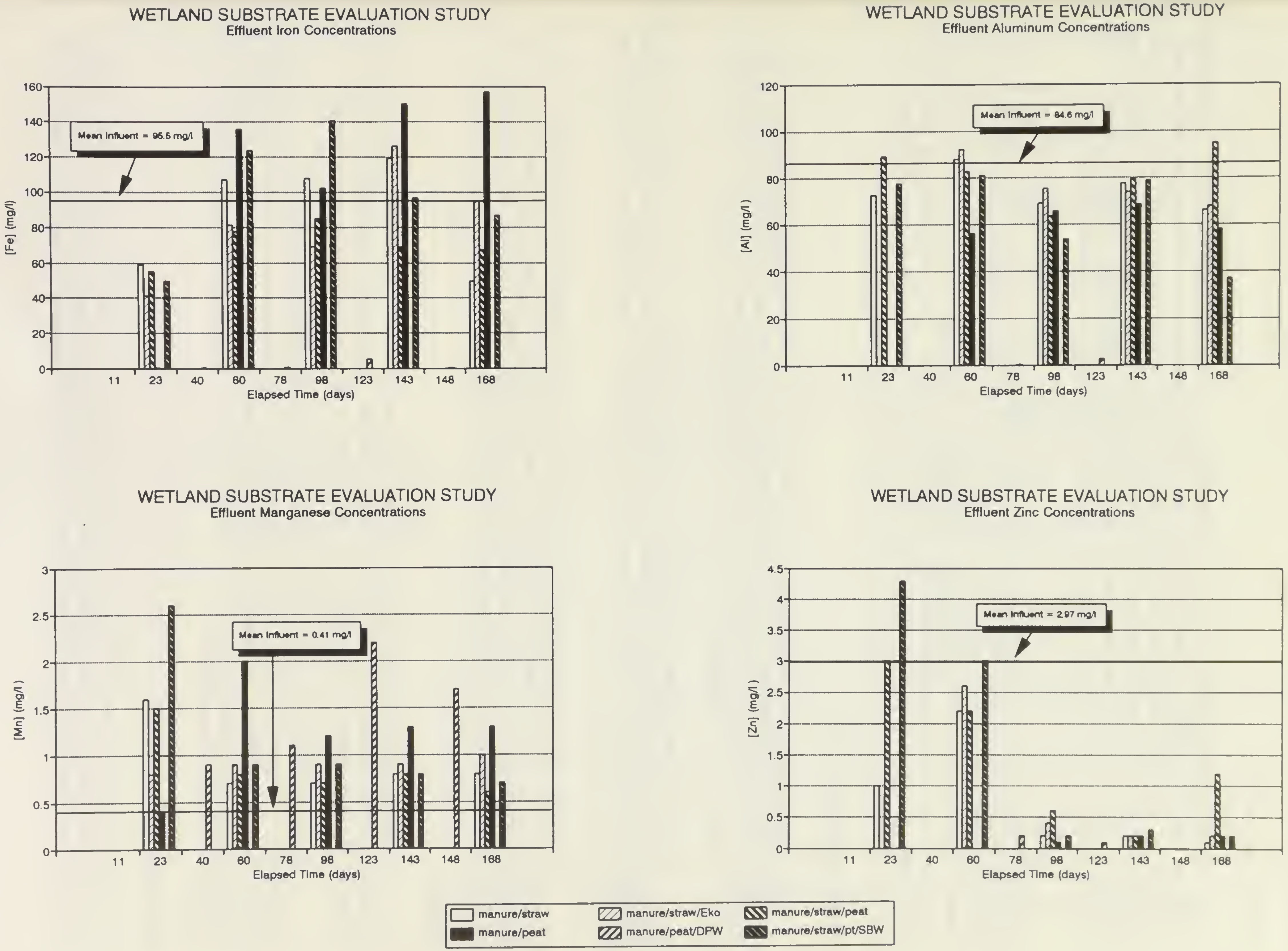

Figure 1. Substrate effluent Fe, Al, Mn and Zn concentrations through time.

Results for Al (Figure 1) were very similar to Fe. Removal efficiency ranged between -12 - 56 % throughout the study for all substrates except manure/peat/DPW, which removed > 97 % of all Al throughout the study.

Manganese and Zinc

Overall, all substrates were net Mn producers (Figure 1). Manganese concentrations were generally highest in the initial sampling, probably due to a flush of Mn in the substrate materials as they were exposed to low pH (< 3.0) AMD. Aluminum and Fe are present in concentrations far greater than Mn and may have displaced Mn from exchange sites by mass action and preferential adsorption of Al and Fe over Mn. The highest Mn concentrations in the latter part of the study are associated with the DPW-amended substrate. This is probably the result of strong reducing conditions, which increase the solubility of free Mn^{+2}.

After an initial flush of Zn (Figure 1) from the manure/straw, manure/straw/peat and manure/straw/peat/SBW substrates, all substrates provided substantial removal of Zn throughout the study (Figure 1). These increases may be due to liberation of Zn present in the substrates by the introduction of the low pH AMD. In the final 3 sampling rounds, all substrates, with the exception of manure/straw/peat, removed > 93 % of influent Zn. This may be a result of limited SRB activity which resulted in the production of reduced S in concentrations adequate to precipitate Zn as ZnS. The precipitation of ZnS is also thermodynamically favored over the precipitation of Fe or Mn monosulfide, having a solubility product 5 and 8 orders of magnitude lower than FeS and MnS, respectively (Ehrlich 1981).

Acidity/Alkalinity and pH

Effluent data indicate that only the manure/peat/DPW mix yielded net alkalinity values (Figure 2) for any significant period of time and showed increasing alkalinity concentrations on subsequent sampling dates. The trend of increasing alkalinity after Day 78 indicates the production of alkalinity in excess of that associated with the inherent NP and suggests SRB activity, a claim corroborated by visual observations and SO_4 data.

Net acidity values (Figure 2) are negatively correlated to alkalinity values and. Acidity levels show a gradual increase through time for all substrates with the exception of the manure/peat/DPW mix, which was a net alkalinity producer, and manure/straw/peat/SBW, which exhibited a gradual decline from Day 60 through the completion of the study.

Generally, pH values decreased through time as acidity increased for all substrates except manure/peat/DPW. Figure 3 shows pH versus time for the manure/peat/DPW and manure/peat substrate. All other substrate pH levels were lower than these 2 substrates. It should be pointed out that small differences in pH between sampling dates is probably due to minor daily perturbations in flow.

Sulfate

Sulfate removal efficiency ranged from 3 to 17 % initially for all substrates excepting the DPW-amended substrate (Figure 4). In subsequent sampling rounds, SO_4 removal ranged between -13 to 7 %. Values of SO_4 were generally at to slightly above mean influent levels, suggesting minimal SRB activity.

Concentrations of SO_4 in DPW-amended substrate was considerably lower (8 to 38 %) than mean influent values. This fact, in conjunction with a trend of increasing alkalinity values and visual observations, suggests the activity of SRB's. Although "black spots", presumed to be FeS were observed to some degree in all columns upon termination of the study and removal of substrate from the columns, they were by far most prevalent in the manure/peat/DPW.

Neutralization Potential

Neutralization Potential (NP) and total acid load (TAL) of each substrate mix are shown in Figure 5. Additionally, the NP of each individual substrate component is shown in Figure 6. The TAL was determined by multiplying the mean influent acidity in mg/l by the total number of liters of AMD which passed

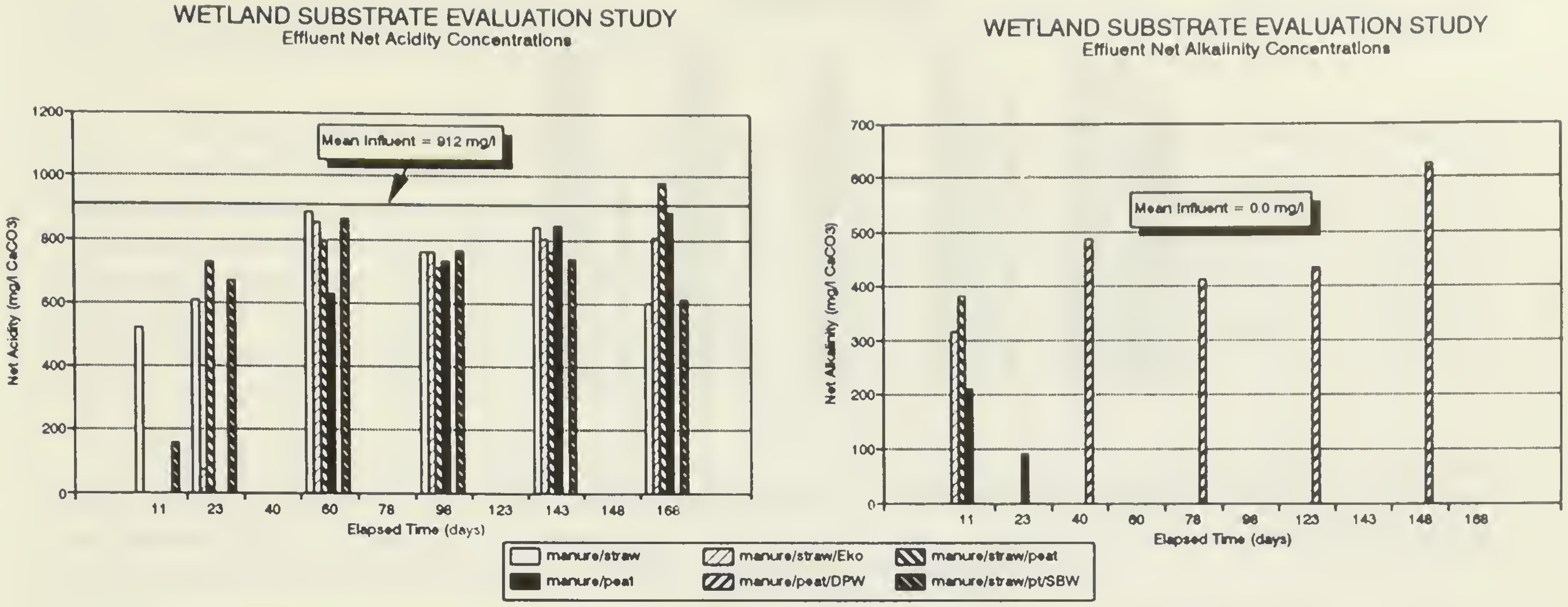

Figure 2. Substrate effluent alkalinity and acidity (as $CaCO_3$) concentrations through time.

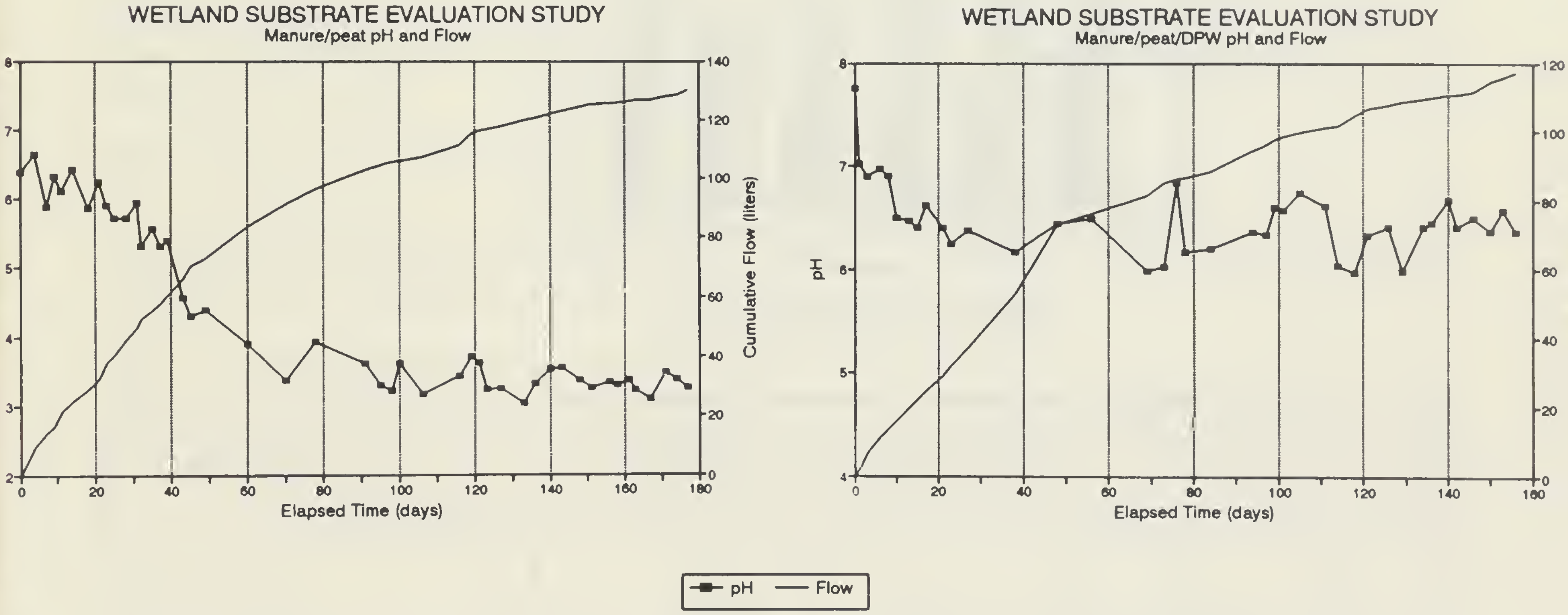

Figure 3. Manure/peat and Manure/peat/DPW effluent pH through time.

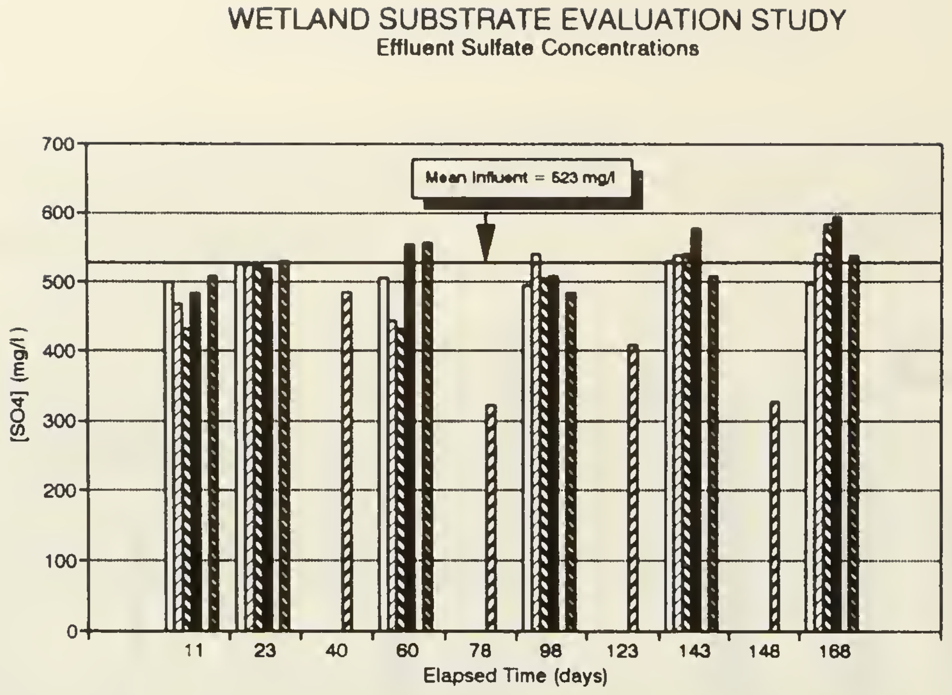

Figure 4. Substrate effluent SO_4 concentrations through time.

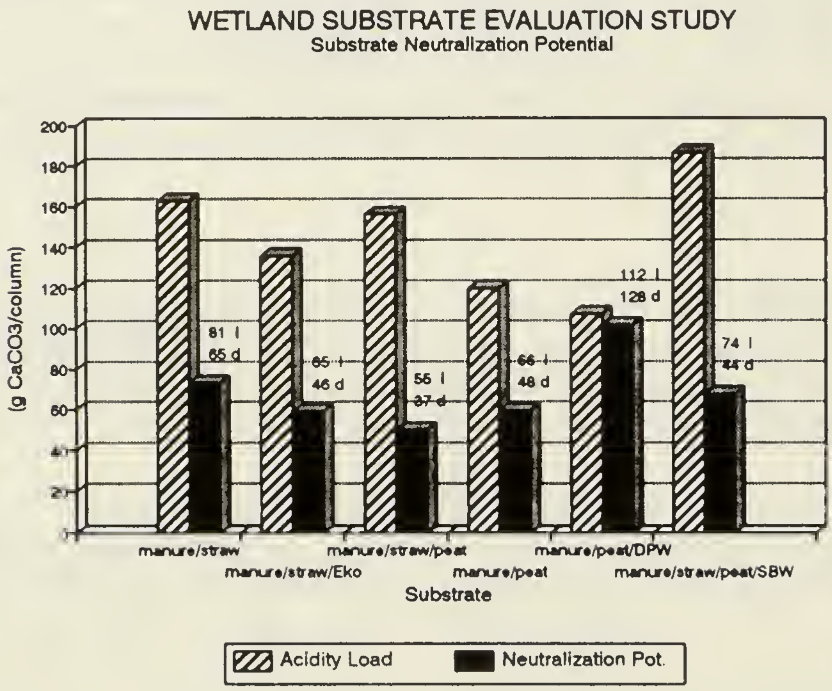

Figure 5. Neutralization potential of substrate mixes.

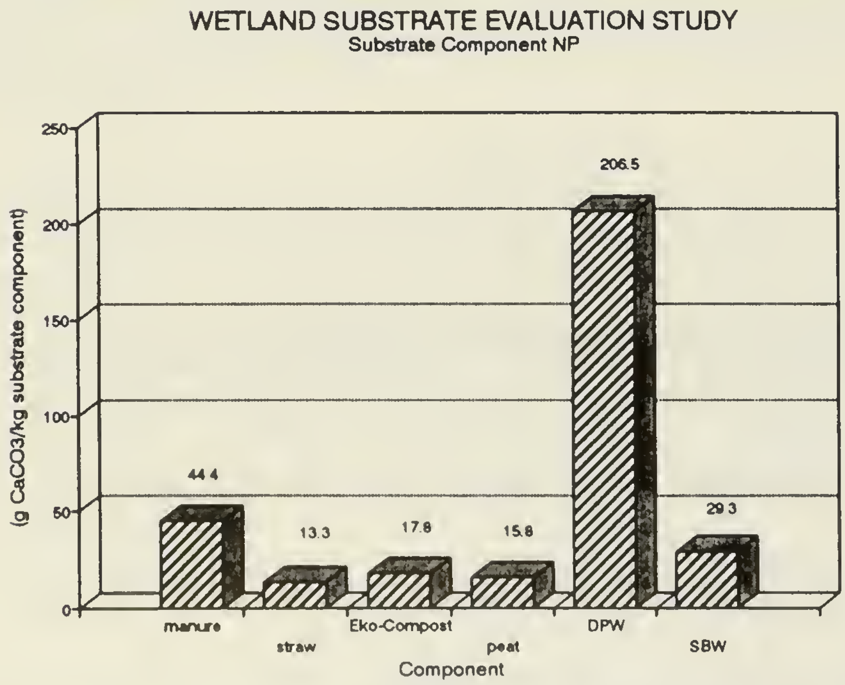

Figure 6. Neutralization potential of individual components of substrate mixes.

in days that it would take to exhaust the inherent NP of the substrate in the column. The number of days until NP exhaustion was calculated using the mean flow rate for each column for the first 66 days of the study. Recall that after this point, flow rates were reduced in half.

The results of NP testing indicate that the substrate mixes containing straw had the lowest NP values, which explains their poor performance at decreasing acidity levels. Manure/straw had a greater NP than the other straw mixes or manure/peat. This is in spite of the fact that straw had the lowest NP of any of the individual substrate components (Figure 6). This discrepancy is likely due to the difficulty in accurately visually determining the volume of straw in a small laboratory (20 gram) mix. The addition of 10 % SBW to manure/peat (at the expense of manure and peat as a percentage of the overall mix) resulted in an increase in NP. However, the increase in NP had no significant effect on the performance of the SBW-amended substrate and would be negligible with respect to long-term performance.

Figure 5 indicates that the NP of all substrates, with the exception manure/peat/DPW, would have been exhausted within 81 days. The fact that exhaustion occurred much earlier in all substrates except manure/peat/DPW is probably explained by the formation of excess acidity associated with hydrolysis and formation of Al and Fe hydroxyoxides.

Figure 6 shows that DPW has a NP of greater than 200 g/kg as $CaCO_3$ (20 % $CaCO_3$) equivalents), or almost 5 times greater than manure and substantially greater than the other substrate components. Thus a large part of the success of the manure/peat/DPW mix at decreasing metals and maintaining alkaline conditions can be attributed to the NP. Figure 5 indicates, however, that the NP of manure/peat/DPW would have been exhausted by Day 128, or 112 liters. The fact that alkaline conditions were maintained beyond this point to the end of the study (Day 157, 118 liters), suggests the contribution of alkalinity from another source, sulfate reduction. Effluent SO_4 data (Figure 4), a temporal increase in alkalinity values and visual observations corroborate this statement.

Effect of Flow Rate on Effluent Chemistry

Flow rates for each column are presented in Table 4.1. Rates are shown for the initial 66 days of the study and for Day 67 through the end of the study, at which point they were reduced approximately in half. Overall mean flow rates and corresponding residence times are also shown.

Table 4.2. Substrate Flow Rates and Residence Times.

SUBSTRATE	FLOW RATE (liters/day)			RESIDENCE TIME (days)	
	Days 0-66	Days 67-177	Overall	Days 0-66	Days 67-177
manure/straw	1.56	0.72	1.05	3.33	7.22
manure/straw/ Eko-Compost	1.42	0.47	0.99	3.66	11.06
manure/straw/ peat	1.48	0.67	0.99	3.51	7.76
manure/peat	1.36	0.37	0.76	3.82	14.05
manure/peat/ DPW[1]	1.52	0.51	0.76	3.42	10.20
manure/straw/ peat/SBW	1.68	0.87	1.19	3.10	5.97

1 - Elapsed time is 20 days less than shown (ie: Days 47-157) due to later start up time.

Flow rates for the Day 0 to Day 66 time period were very similar and ranged between 1.36 and 1.68 liter/day. The discrepancy in values for the Day 67 to Day 177 period is due to the difficulty in maintaining consistent flow at such low flow rates.

The change in flow rate and subsequent increase in residence time produced no significant effects on effluent water quality, with effluent metal concentrations showed little change in response to the decreased rate of flow. It is probable that the sorption capacity, which is one of the primary removal mechanisms for metals, was already saturated or near-saturated by the time flow was reduced. It can also be inferred that the initial flow rate was not limiting (or residence time was adequate) to the formation of metal hydroxyoxides, another important metal removal mechanism, as evidenced by the lack of change in effluent metal concentrations.

There appeared to be no effect of decreased flow rate and increased residence time on SRB activity, as determined by SO_4 levels and visual observations, in any of the substrates, with the possible exception of manure/peat/DPW. This is in spite of the fact that residence times should have provided adequate time for sulfate reduction and metal removal to occur (Kuyucak et al 1991).

Using the maximum mean daily flow rate of 1.68 l/day (manure/straw/peat/SBW) and the mean influent SO_4 concentration (523 mg/l), a SO_4 loading rate of 0.879 g SO_4/l/day or 0.00054 g SO_4/cm^3 substrate/day is obtained. This is well below rates of SO_4 reduction rates of 0.029 to 0.36 g/cm^3 observed by Dvorak et al (1991) and 0.0577 g/cm^3 (McIntire et al (1990) in substrate barrel experiments and a constructed wetland, respectively. It therefore appears that adequate residence time was allowed for sulfate reduction to occur in the columns and that the sulfate reducing process was kinetically but not thermodynamically limited. Although not reported in this paper, levels of TOC and TKN also were not limiting. It therefore is likely that the harsh (low pH, high metals) conditions in the columns seriously inhibited SRB activity and sulfate reduction. Kuyucak et al (1991) found this to be the case using several organic substrates and comparing sulfate reduction rates in neutral water, and synthetic and natural AMD. This claim is underscored by the fact that even following inoculation on Day 121, there was no apparent increase in SRB activity.

Because there was a significant amount of inherent NP present in the manure/peat/DPW mix throughout most of the study, it is difficult to determine the contribution to total alkalinity from sulfate reduction. Results from the manure/peat/DPW do suggest, however, that the initial flow rate was not limiting to the activity of SRB's and that the maintenance of alkaline conditions (and subsequent low metals and neutral pH) permitted the activity of SRB's. The maintenance of circumneutral pH values and an alkaline environment in manure/peat/DPW effluent beyond the exhaustion of the NP corroborates other evidence suggesting the activity of SRB's at a level which was not, at least for the duration of this study, seriously kinetically limited.

DISCUSSION

Potential Use for Substrates in Wetlands

Research by Wieder et al (1985), Wieder et al (1990), Calabrese et al (1991), Henrot and Wieder (1990), Taddeo and Wieder (1991) and Schafer and Associates (1991), indicates that sulfate reduction and the subsequent formation of FeS and FeS_2 does not play a significant role in Fe removal (< 37 % in all cases and < 20 % in most cases), and that adsorption mechanisms and precipitation of hydroxyoxides or carbonates are the major processes for Fe removal from AMD in wetlands. Results of the Phase I study corroborate these findings.

The primary value of sulfate reduction in treating AMD then becomes its role in the production of alkalinity and the maintenance of circumneutral pH values. In a case where sulfate reduction was observed to be occurring, substrate pore water pH values between 6 and 7 were measured by McIntire et al (1990) in the Friendship Hill experimental wetland, which was constructed in Pennsylvania by the USBM to treat AMD. However, surface water pH values were between 2.5 and 3.5. It was believed that if the AMD could be forced through the substrate to maximize contact time, treatment efficiency would greatly improve. This was accomplished through the use of plexiglass "dams" to force water downward and through the substrate. The increased flow through and contact time with the substrate resulted in pore water pH values of between 3.2 and 4.5. This suggests that the majority of pH control and alkalinity in the pore water was the result of the inherent NP of the substrate and not sulfate reduction.

Postgate (1984) showed that below a pH of about 5.0, SRB activity was severely inhibited. High concentrations of metal also inhibit SRB activity (Kuyucak et al 1991). Although research results (McIntire et al 1990, Dvorak et al 1991) indicate that SRB's can remove metals and contribute alkalinity to AMD, the role of SRB's in maintaining alkaline conditions in a wetland is not clearly understood. This is because most results are from short term studies using mushroom compost. Because mushroom compost has a considerable NP, results are usually favorable (low metals, neutral pH). The Phase I study showed very similar results for the mushroom composts evaluated, where alkaline effluent conditions were maintained largely by the inherent NP of the mushroom compost. Calabrese et al (1991) also obtained favorable results using a limestone underlayer, which under reducing conditions did not coat with metal hydroxyoxides and provided substantial alkalinity for the maintenance of an environment conducive to SRB growth. These studies, however, do not accurately depict the long-term potential for AMD treatment and the sustainability of SRB activity in a low pH, high metal environment after exhaustion of substrate NP. When other organic substrates having lower NP values than mushroom compost were used, including manure/straw, peat, peat/limestone/fertilizer, sawdust (Taddeo and Wieder 1991) and manure/peat/wood waste (Howard et al 1989), water quality was typically poorer and SRB activity was considerable inhibited. These findings are corroborated by the results of the Phase I study and this study. Similar performance has also been observed in constructed wetlands.

In summary, it appears that sulfate reduction has been sustainable at fairly high rates in wetland substrate materials which provided enough NP to maintain a circumneutral pH environment conducive to the growth and activity of SRB's. However, the contribution of alkalinity derived from sulfate reduction as opposed to alkalinity inherent in substrate or added limestone has been difficult to determine. Also, where residence times have been seemingly adequate, SRB activity has been inhibited by the acidity and high metal concentrations in AMD. Although the establishment of viable SRB communities and attendant sulfate reduction has been observed in wetland substrates subjected to AMD, this success is likely the result of favorable growth conditions maintained by the inherent NP of the substrate. Following depletion of the NP it is unclear whether sulfate reduction can be maintained at a rate high enough to effectively neutralize acidity and remove metals. Also, specific rates of microbial sulfate reduction for various substrates subjected to AMD conditions has not yet been adequately quantified. Data to this point in time suggest that additional treatment measures need to be implemented in concert with sulfate reducing wetlands to provide adequate treatment of AMD. Although reduction in influent flow volumes may ultimately increase the effectiveness of sulfate reducing wetlands, such reductions may result in an economically infeasible cost:benefit ratio and decrease the utility of wetlands as a long-term, cost-effective AMD treatment alternative.

CONCLUSIONS

Based on the results of column substrate effluent and pore water chemistry, substrate neutralization potential and visual observations, the following conclusions regarding specific substrate mixes can be made.

- All substrates with the exception of manure/peat DPW exhibited poor overall metal removal and pH maintenance performance. Manure/peat and manure/straw/Eko-Compost initially removed the majority of Al and Fe (manure/peat). However, by Day 60 (15 to 19 PV), Al and Fe effluent concentrations were near or greater than influent values and conditions were strongly acid.

- Manure/peat/DPW consistently removed greater than 95 % of all influent Fe and Al and maintained alkaline and circumneutral pH values.

- All substrates consistently reduced Zn concentrations significantly below influent levels after Day 60 (15 to 19 PV). Manure/peat DPW decreased Zn concentrations to below detection limit (0.1 mg/l) throughout the study.

- All substrates were net Mn producers, resulting in effluent Mn concentrations which were greater than influent concentrations throughout the study.

- Effluent SO_4 levels from manure/peat/DPW were considerably lower than concentrations in other substrate effluents. This fact in conjunction with readily visible "black spots" and layers in the manure/peat/DPW corroborate the activity of SRB's and sulfate reduction.

- Influent SO_4 concentrations were below the maximum values for sulfate reduction based on a comparison to values found in the literature, and thus were not excessive compared to observed daily metabolic rates.

- The inherent neutralization potential of each substrate except manure/peat/DPW was exhausted prior to the time predicted based on acidity loading, likely as the result of acidity associated with the initial hydrolysis of metals.

- The replacement of 30 % manure/straw with Eko-Compost and peat resulted in no significant improvement in effluent water quality.

- The addition of 15 % DPW to manure/peat (and corresponding 15 % decrease in manure/peat) resulted in a dramatic improvement in effluent water quality, with essentially complete metal removal with the exception of Mn. Production of alkalinity also was significant and increased temporally. The success of manure/peat/DPW is based largely on its high inherent NP, a result of the addition of DPW, but also to some extent on the alkalinity contributed by sulfate reduction.

- Inoculation of columns with SRB's resulted in no significant improvement in effluent water quality. Inoculation was performed on Day 121 after effluent water quality had significantly degraded (low pH, high metals). These conditions likely inhibited the establishment of SRB's.

- A decrease in flow rate and subsequent increase in residence time, from between 3 to 4 days to 6 and 14 days, did not result in a significant improvement in effluent water quality from any substrate, with the possible exception of manure/peat/DPW.

- Sulfate reduction was very limited in all substrates with the exception of manure/peat/DPW. This was in spite of apparently adequate concentrations of soluble carbon, nitrogen (TOC and TKN) and SO_4.

- The harsh environment (low pH, high metals) of the substrate pore water severely inhibited SRB activity (sulfate reduction). The maintenance of circumneutral pH and alkaline conditions as a result of a high inherent NP provided an environment conducive to the establishment and viability of SRB's and sulfate reduction in manure/straw/peat and was a major factor responsible for its performance.

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Planning, Rehabilitation and Treatment oF Disturbed Lands
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WETLAND RECLAMATION AT A SURFACE COAL MINE IN MONTANA

Bruce C. Waage[1]

ABSTRACT

WETLAND RECLAMATION AT A SURFACE COAL MINE IN MONTANA

The protection of our nation's wetlands is a great national concern. This concern was expressed in a declaration recently made by President George Bush at the sixth International Waterfowl Symposium when he stated that our national goal would be no overall net loss of wetlands. To make good on his pledge, we need as resource managers to control the loss of existing wetlands and to create new wetlands; Cost effective opportunities to create wetlands are few and require great expense and effort. Although the efforts made in the coal mining industry may be small in relation to the scope of the national wetland problem, opportunities do exist and a contribution to wetland creation and reclamation enhancement can result.

Wetland creation at western surface coal mines has historically been overlooked for several reasons. First, the potential for mine spoils polluting surface and ground water has been well documented in eastern coal fields and this concern was the basis for the regulations we have in the west. What was true in the east stimulated the fears that similar impacts may result. Secondly, the restoration philosophy, which is at the root of the regulations, directed mining companies to put the land back like it was. This, by its very nature, limits creative thinking and does not promote creation of wetlands where few existed prior to mining. Therefore, one of the major wildlife and livestock enhancement opportunities for postmine reclamation has largely been overlooked until recently.

This paper will discuss what has been done to create pond and wetlands both pre-law and post-law on mine lands in southeastern Montana.

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NUTRIENT REMOVAL AND PLANT ROOT RESPONSES TO APPLICATIONS OF LIVESTOCK EFFLUENT USING CONSTRUCTED WETLANDS

B.R. Romig[1] and R.W. Tierney[2]

ABSTRACT

A major point source of agricultural pollution stems from untreated wastewater flows originating in feedlots, dairies and locations of concentrated livestock activity. Wastes emanating from these agricultural developments contain nitrogenous species, salts, organic matter and trace metals capable of impacting stream and groundwater systems by lowering available oxygen and contributing to excessive levels of nutrients. A treatment alternative receiving increasing attention is the use of constructed wetlands for the remediation of waste constituents. Wetlands assimilate pollutant as plant nutrient and host microbial activity responsible for the transformation of wastes into gaseous or inert forms. Although it is accepted that constructed wetlands require adapted species for waste modification, little research has been conducted into the plant physiological functions and characteristics that contribute to effective treatment of high organic wastes. The objectives of this study were to ascertain the effect of two liquid dairy waste loading concentrations (plug flow at two week intervals for a two month period) on nutrient removal by four north-temperate wetland species, *Carex nebraskensis*, *Carex rostrata*, *Typha latifolia* and *Scirpus microcarpa*, and to evaluate the role of root aerenchyma (cortical air space responsible for gas exchange in plants undergoing seasonal flooding) as a plant physiological function. Three replicates of each of two concentrations (30% and 70% dilutions of concentrated liquid wastewater) plus planted and unplanted controls were planted in 50 by 18 cm microcosms at the Plant Growth Center, Montana State University. Plants were grown under 16 hour light to 8 hour dark cycles for one month followed by wastewater applications. Pore water sampling from ceramic cups imbedded in the soil medium at three soil depths (15, 30, 45 cm) was done prior to each effluent application. Water was analyzed for pH, total Kjeldahl Nitrogen (TKN), nitrate nitrogen (NO_3-N), and chemical oxygen demand (COD). Plants were destructively sampled and measured for biomass production, root porosity, and total nitrogen. Results showed differences ($p=0.05$) in plant root porosity in response to differing levels of effluent concentration. No differences were observed in TKN at the 30 and 45 cm depths however wide differences were observed at the surface sampling location. Microcosms receiving the low concentration (70% dilution) showed stabilization of nitrogen level over the two month period while units receiving the high concentration (30% dilution) showed increasingly high nitrogen levels. Nitrogen removal efficiency was 70.2% in the low concentration and 56.4% in the high concentration. Removal efficiency of chemical oxygen demand exceeded 94% in both treatments. No correlation was observed between percent root porosity in the wetland species and the level of contaminant removed. Based on these results, wetlands are capable of removing nitrogen and organic constituents from livestock wastewater given some threshold loading rate. Volume of root porosity is not a factor that determines plant efficiency wetland waste modification. Wastewater modification occurs in the surface soil layers lending effective buffering between surface and soil water at depth.

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[1] Graduate Student, Land Rehabilitation Program, Department of Animal and Range Sciences, Montana State University, Bozeman, MT 59717-0290.

[2] Associate Professor, Department of Animal and Range Sciences, Montana State University, Bozeman, MT 59717-0290.

RECLAMATION-MINING OF SULFIDE-RICH MINE DUMPS USING HEAP-LEACH TECHNIQUES: GEOCHEMICAL IMPLICATIONS OF CYANIDE LEACH COLUMN TESTS

Walter H. Ficklin[1], Geoffrey S. Plumlee[1], John H. Bullock, Jr.[1], Maxine Stewart[2]
and Bob Phillips[2]

ABSTRACT

Numerous mine dumps dot the hillsides around the Central City Mining district in central Colorado. The dumps are made up of wall rock material, sulfides of Fe, Cu, Zn, Ag, quartz vein material and sulfide weathering products. Solution Gold, Ltd., of Golden, CO, uses cyanide heap-leaching to extract any remaining gold from some of these mine wastes. As part of their reclamation program, Solution Gold conducts cyanide leach column tests on agglomerated dump material. We characterized the cyanide speciation chemistry of 6 samples of dump material using a column arrangement for a period of 36 or 43 days. The extracted Ag, Cu, and Zn were more concentrated in the test solutions collected in the first few days of the tests, whereas Fe and As were only slightly extracted during the early stages of the experiment but increased steadily in concentration over the course of time. Thiocyanate formed in all column tests but cannot be accounted for solely by cyanide reactions with sulfur released during sulfide oxidation; thus, thiocyanate may have formed by cyanide reactions with intermediate valency sulfur species present in the mine dumps before cyanide leaching. The changes in cyanide speciation and solution chemistry have significant implications for the design and efficiency of mine-dump reclamation efforts.

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[1] U.S. Geological Survey, Box 25046, MS 973, Federal Center, Denver, Colorado 80225-0046.
[2] Solution Gold, Ltd., P.O. Box 1108, Golden, Colorado 80402.

INTRODUCTION

Gold was discovered in the Central City mining district (Figure 1) in central Colorado in 1859. As of 1990, 4,000,000 oz. of gold were produced in the district (Davis and Streufert, 1990); the majority of this production occurred in the late 1800's. There are numerous mine-waste dumps and tailings deposits throughout the region containing a variety of sulfide minerals. Over the years these materials have oxidized and weathered, creating chronic environmental problems, ranging from acid drainage generation to the mobilization of heavy metals and other toxic elements.

Solution Gold, Ltd., Golden, CO, maintains a cyanide heap-leach operation near the Druid Mine (Figure 1) in the Central City district. The material for their heap-leaching operation comes from several nearby mine dumps. Many of these mine dumps have sufficient residual gold concentrations to make heap-leaching a viable operation. In the process the acid-generating sulfides and toxic elements such as copper and zinc are properly contained in the heap when the operation of the heap is no longer active. The mine-dump sites are partially restored to their original condition after the waste material is added to the heap-leach pad.

In order to assess the feasibility of adding mine-dump material to a heap leach, leach column tests are routinely performed by Solution Gold. The tests are made to determine the rate at which gold and silver will be extracted as cyanide complexes and to determine the consumption of cyanide by the mine-dump material. In this study, we characterized the geochemistry and cyanide speciation of solutions collected almost daily during 36-43 day

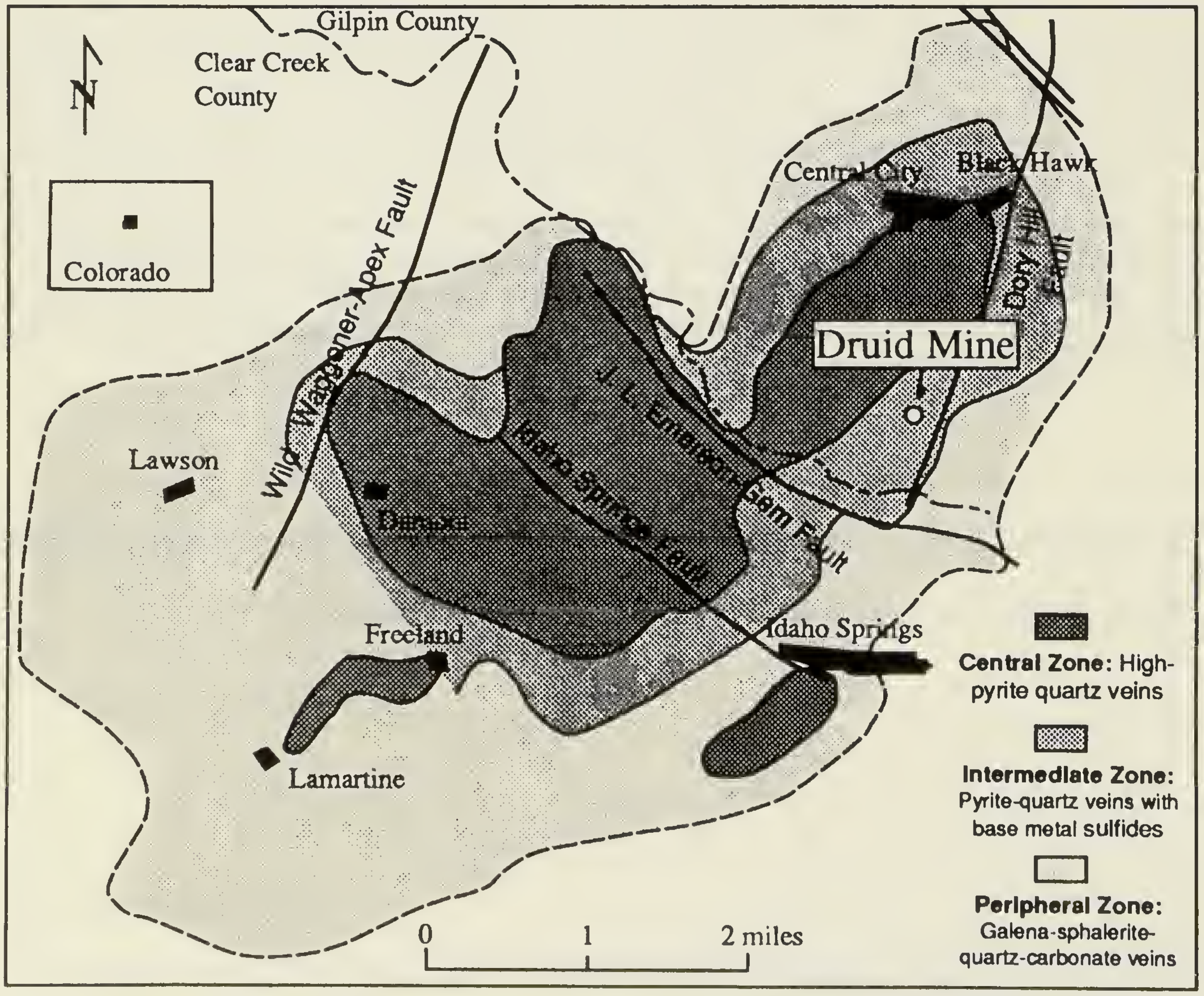

Figure 1. Map of the Central City-Idaho Springs District showing the location of the Druid Mine relative to the district vein mineral zoning patterns. After Sims et al. (1963)

column tests of material from 6 different Central City waste dumps. The results of this study provide important insights into the geochemical processes which control solution geochemistry and metal extraction during cyanide processing of sulfide-bearing mine dumps.

MATERIALS AND METHODS

Leach columns were constructed from plastic (PVC) pipe 6 feet long (192 cm)and six inches (15 cm) in diameter. Each of the six columns (no. 22 to 27) contained 80 pounds (36 kilograms) of mine dump material agglomerated with lime and Portland cement. A 1-pound-per-ton sodium cyanide solution (about 265 ppm CN^-) was dripped through each column at the rate of about 5 L a day. Free cyanide concentrations and pH were determined by Solution Gold personnel on solution samples routinely collected. Splits of 100 to 125 ml were saved for geochemical investigations. Columns 22, 23, and 24 were run for 36 days and columns 254, 26, and 27 were run for 43 days.

Solution samples were stored in polyethylene bottles. Cu, Zn, Ag, and Fe concentrations were determined by flame atomic absorption spectrophotometry. Thiocyanate was determined by forming the red color of thiocyanate iron (III) complex (Standard Methods for the Examination of Water and Wastewater, 1985). Cyanate concentrations were determined by ion chromatography. Arsenic was determined by graphite furnace atomic absorption (Ficklin,1983). A portion of each sample (19 ml) was treated with concentrated nitric acid (1 ml) and allowed to stand for 24 hours to volatilize cyanide; each of these sub samples was analyzed using inductively coupled mass spectrometry (ICPMS) for a qualitative and semi-quantitative determination of 51 elements using an unpublished experimental technique that is under development by the USGS (A.L. Meier, oral commun., 1992). At the conclusion of the leach column tests, samples of the column material were dried. Some metals in this material were determined by six step semi-quantitative emission spectroscopy (Grimes and Marranzino, 1965).

RESULTS AND DISCUSSION

The most abundant sulfides associated with the Central City gold deposits are pyrite (FeS_2), chalcopyrite ($CuFeS_2$), sphalerite (ZnS), and galena (PbS) (Lovering and Goddard, 1950). Less abundant sulfides include enargite (Cu_3AsS_4), tennantite ($Cu_{12}[As,Sb]_4S_{13}$), molybdenite (MoS_2), and acanthite (Ag_2S). Gold occurs as electrum, a native metal alloy of silver and gold. The mine dumps at Central City contain these ore minerals, oxidation products of these minerals, and other gangue minerals such as quartz and silicates. The sulfides still present in the dump material are coarse-grained, up to several cm in diameter. The mineralogy, sulfide grain size, and trace element content of the dumps play a major role in controlling the metals extracted and changes in solution that occur during cyanide leaching of the dump material.

Changes in Solution Chemistry and Cyanide Speciation Over Time

Solution chemistry, metal extraction, and cyanide speciation changed considerably over the course of the column tests; results from column number 27 are shown in Figure 2. In column 27, solution pH (Fig. 2A) dropped substantially in the first 3 days of the test from 11.7 to around 10.8; the pH then decreased slightly through day 15, and then remained relatively constant . The initial pH drop most likely reflects the dissolution of fine-grained agglomerate material. The leveling-off of pH after the first five days of the test indicate that the agglomerated dump material effectively buffered solution pH against acidity increase resulting from sulfide oxidation.

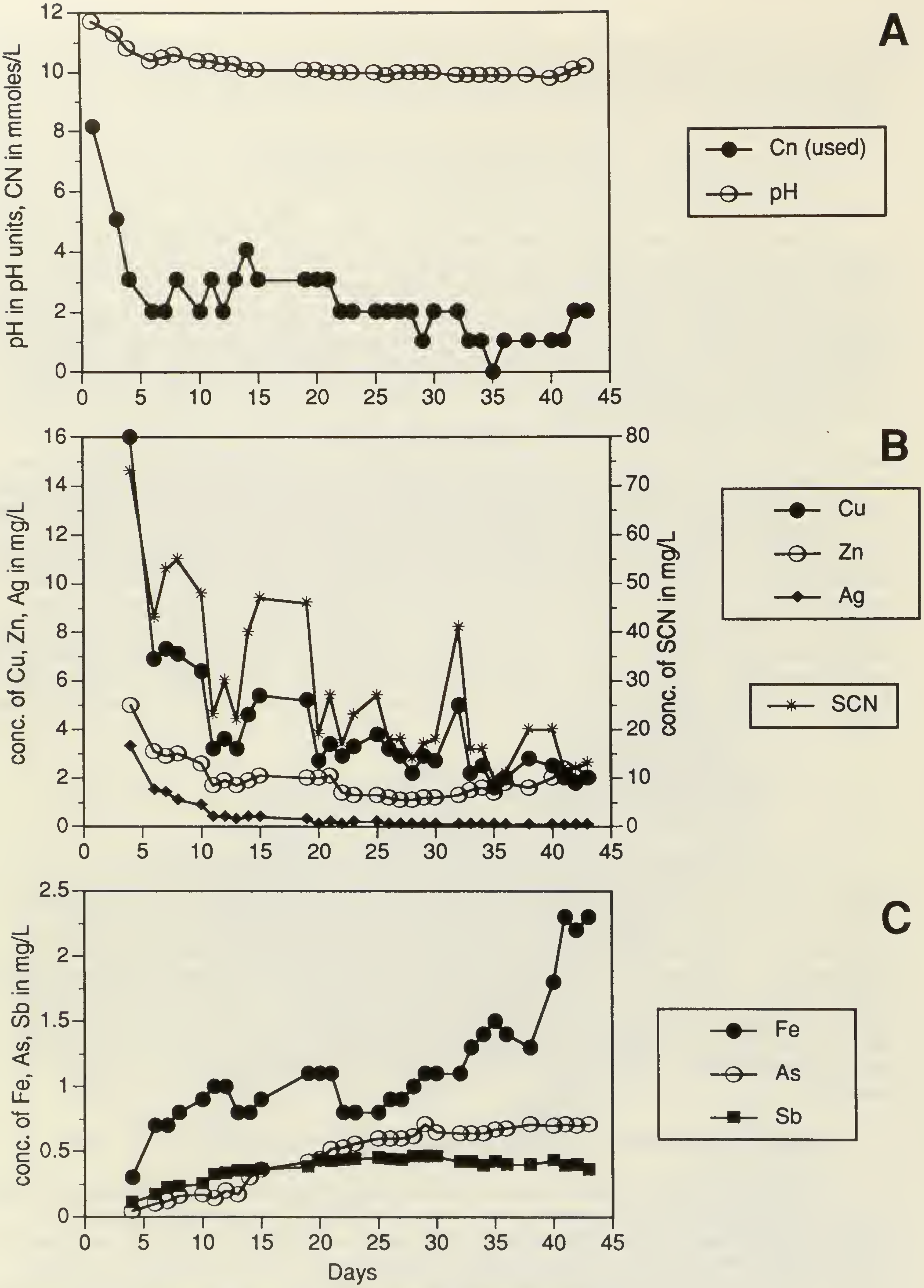

Figure 2. Variations in solution chemistry over time in column 27.

A. Solution pH and amounts of cyanide used (mm/L). The cyanide used is calculated as the starting cyanide concentration minus the free cyanide concentration

B. Concentrations of Cu, Zn, Ag, and SCN in mg/L

C. Concentrations of Fe, As, Sb in mg/L

Extraction of metals such as Zn, Cu, Ag, and Au was greatest in the first several days of the column tests and then decreased substantially (Fig. 2 B). Silver occurs in most weathering environments as an alloy with gold, as native silver, or as salts such as cerargyrite (AgCl); silver in these forms is readily available for complexation by cyanide and should be efficiently extracted early in the column test. Copper and zinc reside in the dumps as both sulfides or oxidation products of sulfides. The high extraction rates early in the tests indicate that the extracted copper and zinc are resident as oxidation products of the sulfides ($CuSO_4$, $ZnSO_4$, and intermediate valency sulfur salts) because the rate of reaction of cyanide with primary copper or zinc sulfides is very slow (McGill et. al., 1985). Likewise, silver still present in the dump material as unoxidized sulfides should show a much lower extraction efficiency than silver present in oxidized material.

In contrast to Zn, Cu, and Ag, extraction of Fe, As, and Sb (Fig. 2C) was negligible early in the tests but increased steadily over time. Iron oxides do not react rapidly with cyanide (American Cyanamid Co., 1950), so the extracted iron most likely is the result of pyrite oxidation. The slow but increasing rates of reaction for arsenic and antimony are also likely due to oxidation of As- or Sb-bearing pyrite or other sulfides. The oxidation products of arsenic are strongly adsorbed by iron oxide (Pierce and Moore, 1982); because iron oxides are not reactive with alkaline cyanide solutions they would likely not release adsorbed arsenic. Very little data is available on the reactions of antimony with cyanide; however, the similarity of the arsenic and antimony extraction curves suggests that arsenic and antimony are responding to the cyanide treatment in a similar fashion.

Thiocyanate Geochemistry

Thiocyanate forms when sulfide minerals react with cyanide solutions. McGill et. al. (1985) found that thiocyanate formation during sulfide oxidation requires 20 days for 10% formation of SCN from $CuFeS_2$ and stoichiometric amounts of cyanide . Experiments with the minerals FeS_2, ZnS, and elemental S produced even smaller concentrations of SCN. In contrast, Luthy and Bruce (1979) found that thiocyanate formation occurred in minutes when cyanide was reacted with polysulfides or thionates, intermediate sulfur species which occur during the oxidation of sulfide to sulfate. Thiocyanate formation from sulfate is not likely to occur in alkaline cyanide media. Thus, formation of intermediate-valency sulfur species may be an important factor controlling the rate of thiocyanate formation during sulfide mineral oxidation.

Elution profiles for thiocyanate are similar to those for Cu, Zn, and Ag (Fig. 2B). This may indicate that the metals are being extracted as a result of sulfide oxidation, and thiocyanate is formed by reactions of the cyanide solutions with sulfur released through sulfide oxidation. However, in all columns, the measured thiocyanate concentrations are far in excess of the theoretical concentrations calculated assuming that Cu, Zn, Fe, and thiocyanate sulfur in solution were derived solely from chalcopyrite, sphalerite, and pyrite oxidation (Figure 3). These data show that neither the sulfur required to form thiocyanate nor the Zn and Cu in solution originated directly from the oxidation of residual sulfide minerals in the mine dumps. The thiocyanate may form through reactions of cyanide with intermediate-valency sulfur species present in the original dump material (possibly contained in soluble salts?). Alternatively, we are currently evaluating whether copper or zinc in solution could have catalyzed the formation of thiocyanate from aqueous sulfate.

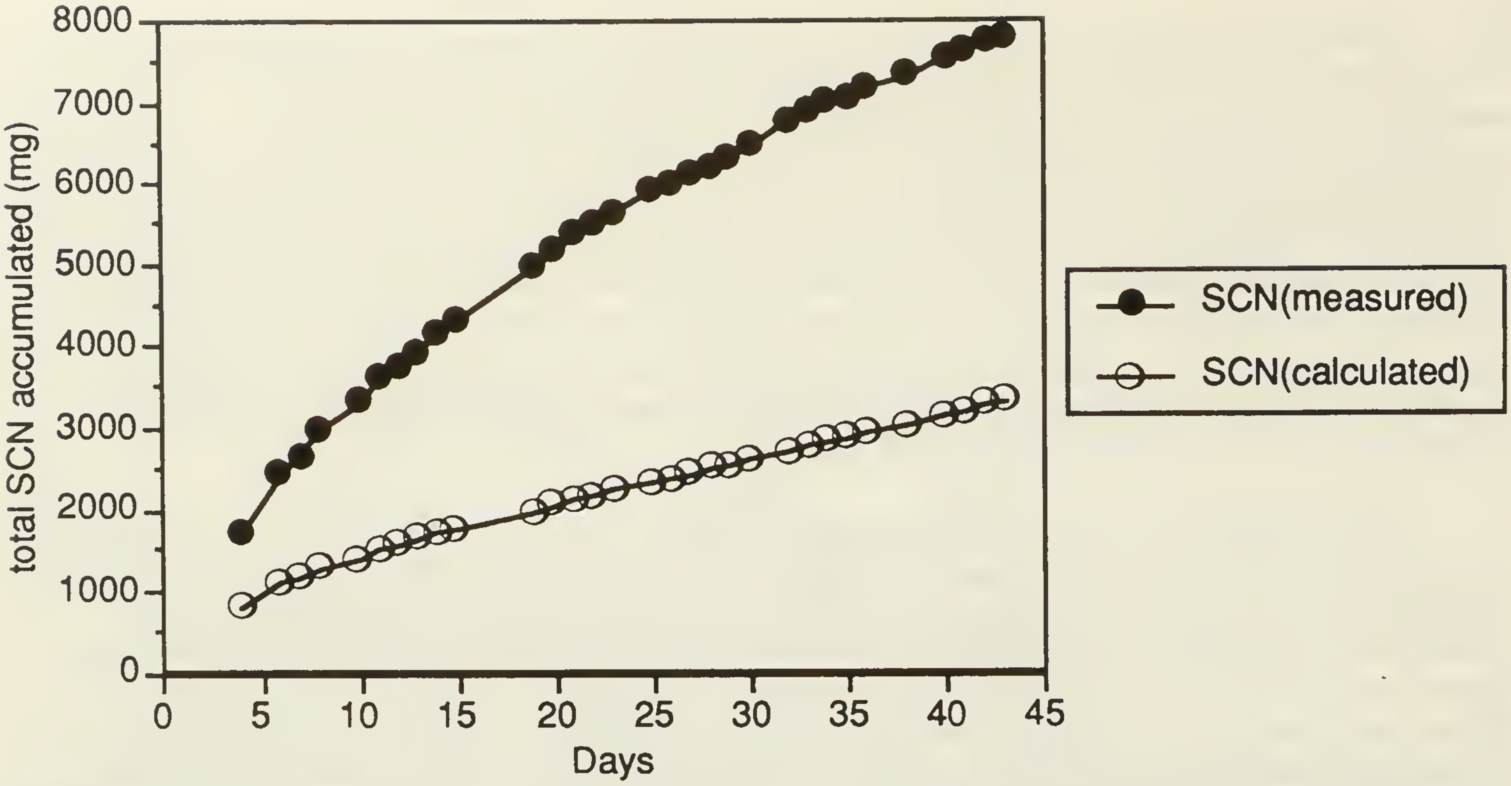

Figure 3. Plot showing measured SCN (measured) and theoretical (SCN calculated) concentrations of thiocyanate in column test 27. The theoretical thiocyanate concentrations are those that would occur if all of the metals present in solution (Table 1) originated from the oxidation of sulfides and all the sulfur released during sulfide oxidation were converted to thiocyanate.

Metal Extraction Efficiency

The total amounts of various metals extracted and cyanide consumed during the leach-column tests are shown in Tables 1 and 2. Some of the metals listed in Table 1 are those that would be expected to occur in the test solutions because of the mine-dump mineralogy. Some less abundant elements determined from the ICPMS scan are included in Table 2. These data can be used to assess the metal extraction efficiency of the cyanide solutions.

$CN_{(m)}$ in Table 1 represents the amount of cyanide theoretically required to form the dominant complexes $Cu(CN)_4^{3-}$, $Zn(CN)_4^{2-}$, $Fe(CN)_6^{4-}$or $Fe(CN)_6^{3-}$, and $Ag(CN)_2^-$ from the amounts of these metals listed in Table 1. CN is the amount of consumed cyanide that is not part of the cyanide used to form thiocyanate and cyanate. CN was calculated using the following equations:

$$1. CN_{(used)}=CN_{(start)}-CN_{(free)}$$

$$2. CN=CN_{(used)}-CN_{(s)}-CN_{(o)}$$

where

$CN_{(used)}$ = amount of cyanide that was used in the leach column test
$CN_{(start)}$ = 1 pound/ton =265 ppm=10.2 mm/L
$CN_{(free)}$= uncomplexed cyanide determined by analysis
CN = cyanide used that was not part of thiocyanate or cyanate
$CN_{(s)}$= cyanide contained in measured thiocyanate
$CN_{(o)}$= cyanide contained in measured cyanate.

If the use of cyanide were limited to the formation of thiocyanate, cyanate, and metal complexes, the amounts in the CN and $CN_{(m)}$ columns should be the same. For columns 24 and 25, in which zinc is the most abundant metal, the actual cyanide used is far less than the theoretical amount required. This suggests that zinc may not be entirely complexed as $Zn(CN)_4^{2-}$. For the other columns, the consumption of cyanide is greater than or nearly equal to the concentration required to complex the metals. The excess consumption of cyanide suggests that some metal cyanide complexes may be retained on the column and that some of the cyanide is lost because of other reactions such as volatilization as hydrogen cyanide or oxidation to carbon dioxide and ammonia.

Table 1. Total amounts of major metals and arsenic extracted (in milligrams), and cyanide consumed (in millimoles) over the course of the leach-column tests. CN and $CN_{(m)}$ are explained in the text. Liters are the total volume of leach solution that passed through columns 22, 23, and 24 in 36 days and columns 25, 26, and 27 in 43 days. nd—not detected.

Column	Cu	Zn	Fe	Ag	As	SCN	CN	CN(m)	Liters
	mg	mg	mg	mg	mg	mm	mm	mm	
22	1400	910	90	1410	4.8	190	164	178	168
23	1460	910	78	1520	2.4	201	251	187	167
24	830	15400	100	130	nd	226	597	1027	246
25	360	2400	150	140	2.3	121	139	185	205
26	1330	590	250	130	69	192	509	153	236
27	1220	550	275	150	110	130	400	140	255

Table 2. Total amounts of less common metals extracted (in milligrams) over the course of the leach-column tests. Determined by ICPMS.

Column	Co	Ni	Cr	Mo	Cd	Sb	Hg
	mg	mg	mg	mg	mg	mg	mg
22	8.4	21	3.4	46	5.9	34	41
23	6.8	24	3.1	47	6.3	31	32
24	41	85	4.3	16	16	3.3	6.6
25	19	41	6.0	14	14	5.2	3.9
26	16	73	4.2	18	3.4	23	11.1
27	13	54	2.6	15	2.0	85	7.7

The results in Tables 1 and 2 show that a variety of elements other than Cu, Zn, Fe, Ag, and Au were extracted by the cyanide solutions. The metals Co, Ni, Cr, Cd, and Hg likely were extracted through the formation of complexes with cyanide. In contrast, Mo, As, and Sb form oxyanion complexes, and so their extraction during sulfide oxidation was likely enhanced by the high pH of the cyanide solutions (Ficklin et al., 1992).

The results for emission spectroscopic (e-spec) analysis of the column material after leaching are presented in Table 3. The main metals other than iron that remain in the mine dump material are lead, copper, and to a lesser extent, zinc; the e-spec analytical method is not sufficiently sensitive to determine the concentration of residual arsenic or antimony and is only borderline adequate for zinc. Comparison of the amounts of metals left with the metals extracted indicate that there are significant quantities of metals remaining in the column material after cyanide leaching. These metals are most likely present as primary sulfides which did not react with the cyanide solutions. Thus, significant sulfide material will likely remain in the heap leach after gold extraction, and should be taken into account in long-term heap reclamation plans. We are currently carrying out detailed chemical and scanning electron microscope studies of the column test material in order to more accurately determine mode of occurrence of the metals.

Table 3. Amounts of metals remaining in column material after cyanide leaching. Determined via emission spectrographic analysis. Also shown for comparison are the amounts of copper, zinc, and silver extracted, recalculated from Table 1 as ppm.

Column	Cu remaining ppm	Cu extracted ppm	Zn ppm	Zn extracted ppm	Pb ppm	Ag ppm	Ag extracted ppm
22	200	39	<200	25	3000	7	.19
23	500	40	<200	25	3000	20	.55
24	200	23	200	427	1000	3	.08
25	100	6	<200	67	1000	3	.08
26	200	37	<200	16	500	10	.28
27	300	21	<200	11	200	2	.05

One of the most important metals remaining in the mine dump material is Pb. Lead in the samples originated as galena (PbS) but lead was not dissolved by the cyanide solutions used in the leach column tests (all ICPMS lead concentrations were less than 10 ug/L). Any lead that may have dissolved in the slightly alkaline leach solution would have precipitated as lead carbonate or sulfate (the water used to make the cyanide solution contained 1600 mg/L sulfate). However, galena may serve as a source of polysulfide or polythionates that could form thiocyanate. This would help to account for the sulfide that was needed to constitute the measured thiocyanate.

CONCLUSIONS

In order to carefully assess some of the trends and observations presented above, additional laboratory work is required. However the results of the chemical analysis present some interesting trends and provide the basis for much additional research on the reactions of cyanide and the role of cyanide complexes in the environment. Additional work in progress includes more detailed chemical analysis of the leach column material by more sensitive instrumental methods and additional leach column tests on well characterized mine dumps under more carefully controlled laboratory conditions.

Copper, zinc, and several other elements occur in the mine dumps as sulfides and as oxidation products of sulfides. The sulfide oxidization products are readily available for attack by alkaline cyanide solutions, and therefore provide the main source for metals such as Cu, Zn, Ag, and Au which were extracted most effectively early in the column tests. In contrast, Fe, As and Sb were not readily available for reaction with the alkaline cyanide solutions, but were extracted in increasing amounts over the course of the column tests. Apparently the oxidation products of iron, arsenic and antimony do not dissolve in cyanide solutions; instead, oxidation of sulfides apparently provided the source for these metals.

The data show that thiocyanate formation is a major sink for cyanide, and should be of concern in the planning of mine-dump remediation schemes using heap leach techniques. Thiocyanate apparently does not form directly by oxidation of sulfides in the dump material. Instead, it appears that the thiocyanate may form by reactions of cyanide with intermediate-valency sulfur salts present in the dump material prior to leaching. Alternatively, the thiocyanate might form by metal-catalyzed reactions with sulfate in the cyanide solutions. We are currently carrying out studies to assess the relative importance of these two processes.

Finally, our results show that significant amounts of metals, such as Cu, Zn, Ag, and Pb will likely remain in the agglomerated mine dump material after heap leaching has extracted the gold. It is likely that, due to the sulfide-rich, coarse-grain nature of the ore, that these metals reside in sulfides that escaped oxidation by the cyanide solutions. These sulfides could serve as a long-term source of acidity and metals, and must therefore be taken into account in plans for long-term heap management.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

FIELD EVALUATION OF LAND APPLICATION PERFORMANCE: METALS REMOVAL FROM BARREN LEACH SOLUTION[1]

Steven C. Smith, Thomas J. Hudson, and William M. Schafer[2]

ABSTRACT

The Pegasus Beal Mountain Mine has been operating since 1988 as a heap-leach gold producer. The mining plan calls for use of treatment and land application of process solutions as part of facility closure. This field investigation was conducted to evaluate the performance of natural soils in the designated Land Application areas in attenuating metals and residual cyanide contained in treated solution. Soils were instrumented with neutron probe access tubes to monitor changes in water content during solution application and suction lysimeters to collect samples of soil solution from various soil depths. Soils were irrigated with well water during an initial test to measure the chemistry of natural soil solution. Next treated process solution was applied to measure metal and cyanide attenuation. Treated solution contained 3 mg/l total cyanide, 2.3 mg/l WAD cyanide, 0.007 mg/l arsenic, 42.0 mg/l copper, 0.60 mg/l iron, 0.09 mg/l manganese, 5.02 mg/l nickel, 0.612 mg/l selenium, and 0.14 mg/l zinc. Initially, when solution was applied at the rate of 0.6 inches per hour, no runoff was detected and applied water moved rapidly downward into the soil profile. However, one site failed to maintain a percolation rate that would permit sustained application at this rate if continued longer than the actual test. Nonetheless, during the land application test in September, 1991, over 15 inches of treated solution was added before saturated conditions formed at depth. Soil attenuation of constituents contained in applied barren solution was evident at both LAD sites. More than 99% of applied copper and nickel was adsorbed by the soil system. Removal of roughly 90% of applied selenium was evident after solution had infiltrated through 36 inches of soil. Zinc and arsenic removal were not as pronounced. However, these constituents were not present at high concentrations in the treated barren solution. Levels of WAD cyanide were also substantially reduced due to soil attenuation or chemical precipitation reactions. Cyanide in soil moisture also shows a tendency for continued slow reduction in the soil environment after application. Manganese levels in soil pore water were higher than in applied waters for both the fresh-water rinse as well as the barren solution application. Overall, Land Application at the Beal Mountain site appears to be an effective means of polishing the water quality of treated process solution and is a practical alternative to surface discharge of treated solution.

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[1] - We would like to acknowledge Pegasus-Beal Mountain Mine for project funding

[2] - All authors with Schafer and Associates, Bozeman, MT 59715

INTRODUCTION

A number of mines in the western U.S. utilize a "heap leach" method for recovery of disseminated gold from oxidized ore material. The low cost inherent in heap-leaching allows mining of low grade ores. To extract gold, a solution of cyanide-containing solution is applied by sprinkler or subsurface irrigation to the surface of a pile of permeable ore. The circulation of process solution through the ore is maintained by using coarse ore material or by agglomerating fine ore materials. Gold bearing "pregnant" solution is collected along the lined base of the heap leach pad and drains into a pond for subsequent gold recovery and regeneration of cyanide leach solution. The processed "barren" solution is re-used on the heap.

The heap leach system is exposed to water gains and losses through precipitation and evaporation. Many of the earliest heap leach operations were located in Nevada where water loss was substantial, creating a need for "make-up" water. In Montana, water surpluses may develop during certain seasons of operation. Hence, a need for disposal of process solution exists.

The Pegasus Beal Mountain gold mine was permitted in 1988 for operation of a mine and cyanide heap leaching circuit. Permit provisions call for disposal of spent cyanide solution at the end of mine life by a combination of volatilization, photo-oxidation, peroxide oxidation, and finally land application disposal (LAD). Two areas near the mine were permitted as LAD sites. Peroxide treatment and LAD would also be used as a last alternative in an extraordinarily wet year to deal with accumulating water inventory in the process ponds. The probability of such a requirement is very remote, however. The facility is capable of holding the maximum total annual rainfall plus snowmelt probable in a 100 year period.

The study objective was to conduct field application tests using treated process water from the operational facility to confirm the performance of the LAD system incorporating recommended methods as required.

FIELD INVESTIGATION

Approach

Land application is a commonly-used method for disposal and treatment of low toxicity industrial wastewater. Most soils have a strong affinity for heavy metals and other contaminants. A well designed LAD system should result in metals attenuation and removal of residual weakly associated cyanide by attenuation, chemical decomposition or precipitation as insoluble metal cyanide complexes.

Two LAD sites are permitted at the Beal Mountain mine. Site 1 soils are classified as Typic Cryoboroll loamy-skeletal, gravelly silt loams. The aspect is typically east to southeast with an elevation range from 7400 to 7800 feet. Slope was 23 percent at the Site 1 descriptive soil pit and is similar across the site. Site 2 soils are classified as Typic Cryoboroll loamy-skeletal (shallow), gravelly silt loams. Aspect is south to southwest with an elevation range of 7600 to 7800 feet. Fractured bedrock was contacted at approximately 24 inches at both sites.

On both permitted LAD sites, a study area was developed by installing monitoring equipment capable of determining soil moisture content by depth and providing *in situ* pore water samples. Following equipment installation, a fresh water rinse was applied to test the irrigation system and to determine natural background pore water characteristics. Barren process solution was treated with peroxide to destroy free and weak acid dissociable (WAD) cyanide, and reduce metals concentration by precipitation. Treated barren solution was applied onto the two permitted areas by a sprinkler irrigation system similar to that used on leach pads. Soil moisture content was periodically measured to track the initial wetting front and of applied solution and subsequent changes in soil moisture content. Pore water samples enabled metals attenuation and cyanide degradation to be evaluated.

Study Area Design

Each study area consisted of a neutron probe access tube surrounded by a nest of suction lysimeters. Neutron probe access tubes were designed to be installed to the 52 inch depth. Actual depths were 52 inches at Site 1 and 42 inches at Site 2. A *Campbell-Pacific* hydroprobe was used to measure soil moisture content at 15 cm depth increments throughout the soil profiles. Moisture content measurement enabled the wetting front of applied fresh water, or treated barren to be monitored while percolating through the soil.

Soil Moisture, Model 1920 suction lysimeters were installed in a nest around the neutron access tubes. Each site had three lysimeters located approximately at eight feet from the access tube. At Site 1, lysimeters were installed at 16, 36, and 48 inch depths. At Site 2, installations were 16, 30, and 48 inches. Depth variation between 30 and 36 inch lysimeters was due to the presence of indurated bedrock at Site 2 which necessitated termination of the borehole short of the target depth.

Fresh Water Application

During June of 1991, the fresh water rinse was conducted on the LAD sites. The objective of this rinse was to obtain information on water movement through the soil profiles, and to determine baseline pore water chemical characteristics at the LAD sites. A sprinkler irrigation system, which would later be used to apply treated barren was developed and utilized to apply the fresh water. The irrigation rate was set at 2 gpm. Wobbler heads (number 11) that would cover a 20 foot diameter area at the prescribed flow rate were used. This produced an application rate of approximately 0.6 inches per hour. A total of 3800 gallons were applied to both sites, 1900 gallons each. The application rate of 0.6 inches per hour was equivalent to 10 inches of applied water.

H_2O_2 Treatment of Barren Solution

Peroxide treatment of approximately 20,000 gallons of barren leach solution was conducted by Beal Mountain Mining staff. Prior to treatment, barren solutions at the mine typically contain 1 pound CN^- ($AgNO_3$ titratable) per ton of solution, and 400 ppm Cu.

The amount of peroxide used to treat the barren solution was based on testing done by Degussa Corporation which recommended 72 pounds of 50% H_2O_2 per 1000 gallons of barren solution. This level of treatment is approximately three times the stoichiometric requirement but was necessary for adequate cyanide destruction in the current operating leach solution. However, anticipated cyanide concentration at closure will be lower due to volatilization and photo-oxidation and peroxide requirements will be much lower. Prior to land application, samples of the treated barren were taken and submitted for analytical analysis. Table 1 presents the results of the analysis. Copper and nickel remained quite high in the treated barren compared to earlier laboratory tests. This is attributed to the low final pH of the solution. The pH decreased from 10.1 to 7.9 at the time of application. Typically, the pH remained above 9 in laboratory tests. The decrease in pH is caused by high concentrations of soluble metal cyanide complexes, mostly copper cyanide. As cyanide is destroyed the free metals find carbonate and hydroxide ions and precipitate them as insoluble salts. The reduction of free alkalinity causes the pH to decrease.

Treated Barren Solution Application

Application of treated barren solution began on 9/25/91. The rate of application was the same as for the fresh water rinse, at 0.6 inches per hour. Treated barren was supplied to each LAD site by 2 inch supply lines coming from a 3000 gallon storage tank. Supply lines were fitted with number 11 wobbler (sprinkler) heads centered in the study areas. A twenty foot diameter area of coverage was maintained by this irrigation system providing saturation over the areas fitted with monitoring equipment. Application of treated barren continued for 25 hours with a total of 6000 gallons (3000 gallons to each site) being applied. The total solution application was approximately 15 inches of water. Application was discontinuous over a four day period to allow for refilling of the storage tank.

Table 1. Chemical characteristics of treated barren solution at Beal Mountain.

METAL ANALYSES (mg/l)						
As	Cu	Fe	Mn	Ni	Se	Zn
0.007	42.0	0.60	0.09	5.02	0.612	0.14

CYANIDE ANALYSES (mg/l)			MAJOR CATION ANALYSES (mg/l)		
Total	WAD	Free	Ca	Mg	Na
3.0[1]	2.3[1]	<0.2	482	<1	717

[1] Cyanide analysis done by manual distillation, method ASTM D2036.

Data Collection

Soil moisture content determination was conducted prior to, during, and after both the fresh water and treated barren solution applications. Pore water samples were taken at each lysimeter depth immediately following the passage of the wetting front. This first set of lysimeter samples is referred to in the following discussions as initial samples. Pore water was sampled a second time several days following passage of the wetting front. These follow-up samples are referred to as delayed samples. To obtain adequate sample volume, lysimeters were typically purged several times to produce composite samples.

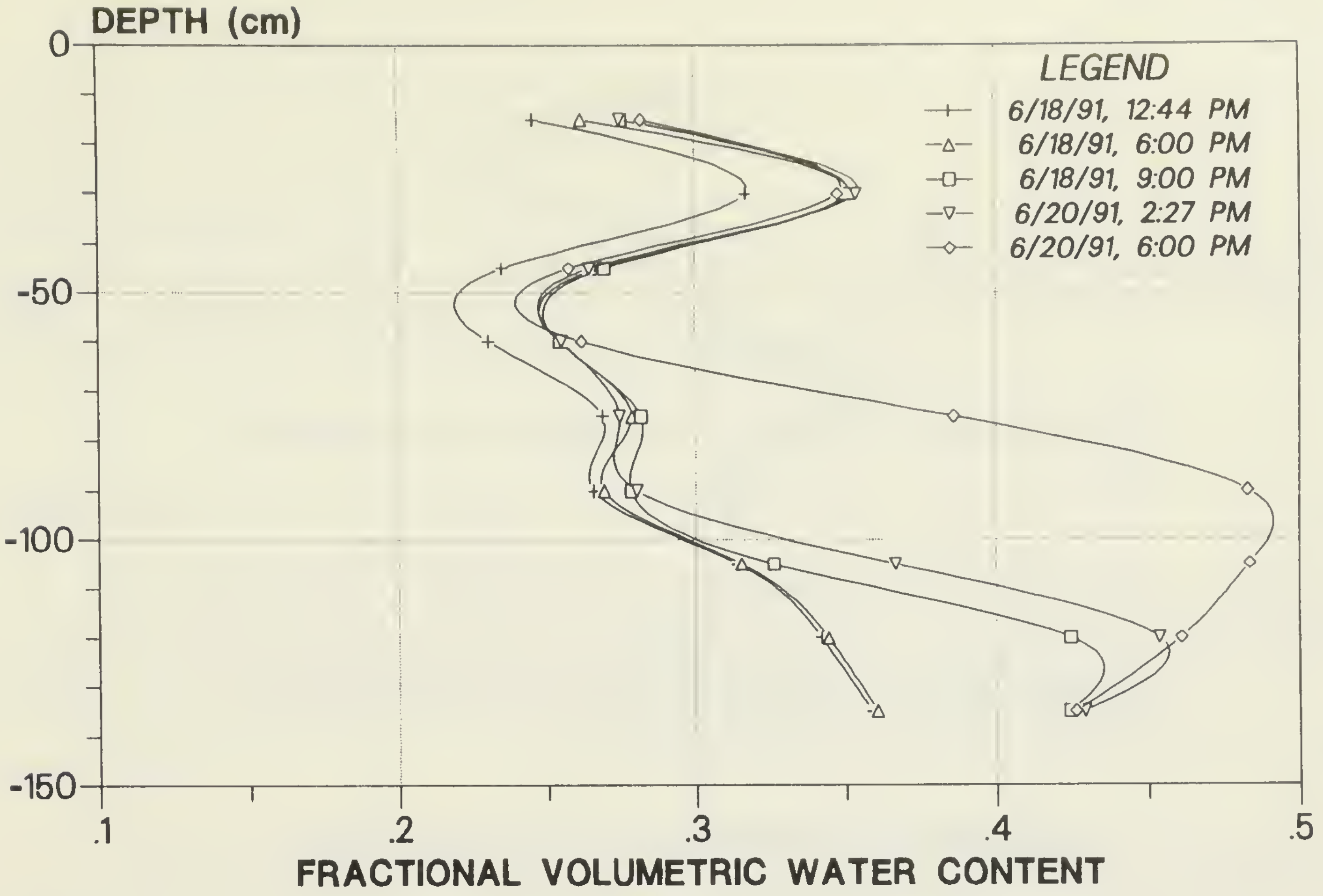
DATA ANALYSIS AND DISCUSSION

Pore water samples were analyzed by Energy Laboratories, Billings, MT using methods approved by ASTM and U.S. EPA. Total and weak acid dissociable (WAD) cyanide analyses were performed by manual distillation (ASTM D2036). Soluble metals analysis was performed by the ICP method.

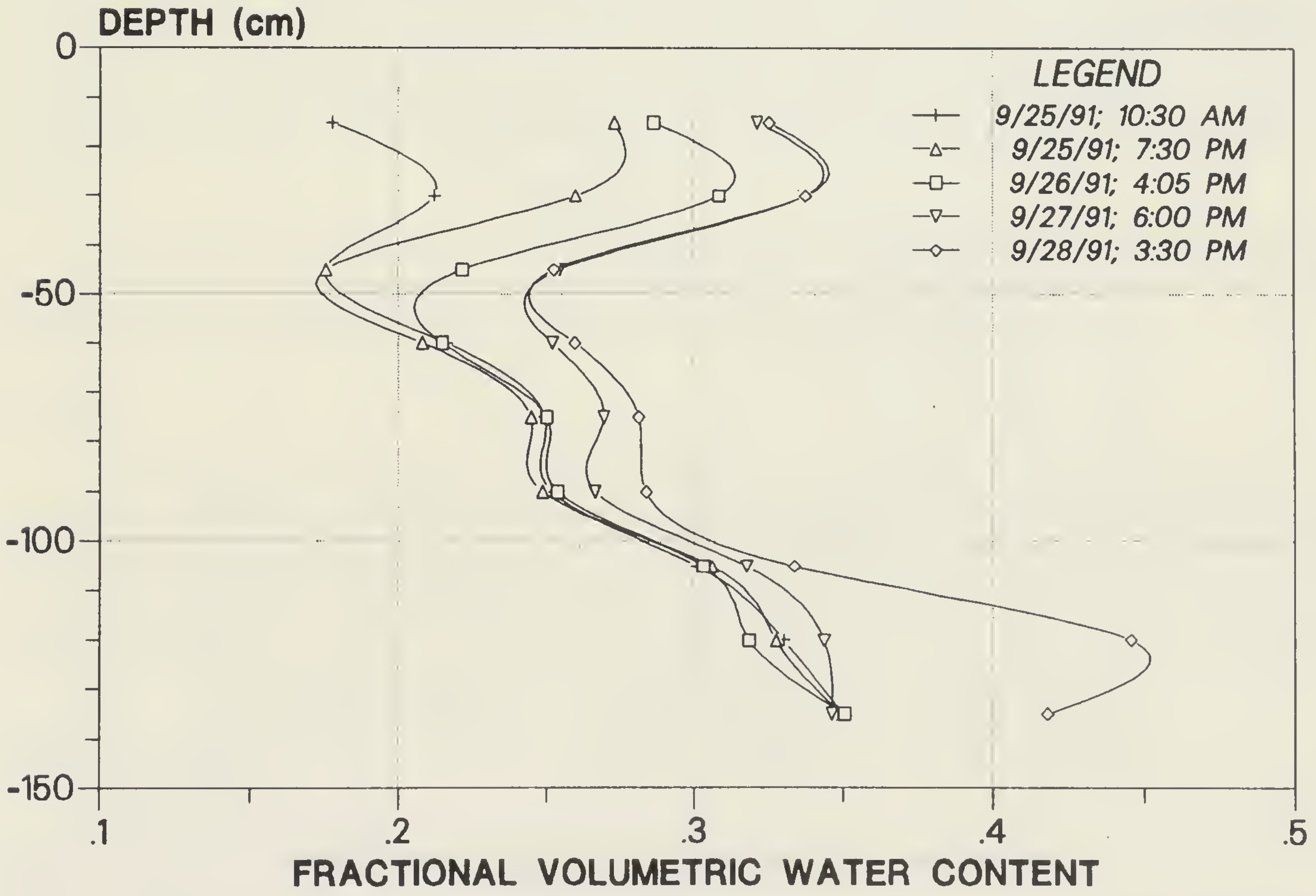
Soil Moisture Content

During the application of fresh water rinse and treated barren solution, soil moisture content was periodically monitored to determine wetting front movement and draindown. Soil moisture data for the two sites are shown in Figures 1 and 2, respectively. Initial moisture contents at both sites were markedly different between fresh water rinse and treated barren applications. This reflects wetter background conditions in June compared to dryer conditions in September. As a result of dry conditions in September, considerably more treated barren was required to saturate the soil system compared to the fresh water rinse conducted in June. In addition, more time was required in September to move a wetting front through the soil profile. Solution retention time in the soil system would be increased, at least initially, if application were conducted under dry soil conditions typical of late summer and fall seasons.

Site 1 appeared to be prone to subsurface flooding. After several hours of water application and movement under unsaturated conditions, water content began to increase markedly from the bottom upwards. It appears that water reaches a relatively impermeable strata at a depth not far below the monitoring instruments and begins to form a perched water table. This may require more frequent rest periods or lower application rates to prevent surface runoff from this LAD area. Water levels at this site were observed to drop if left overnight.

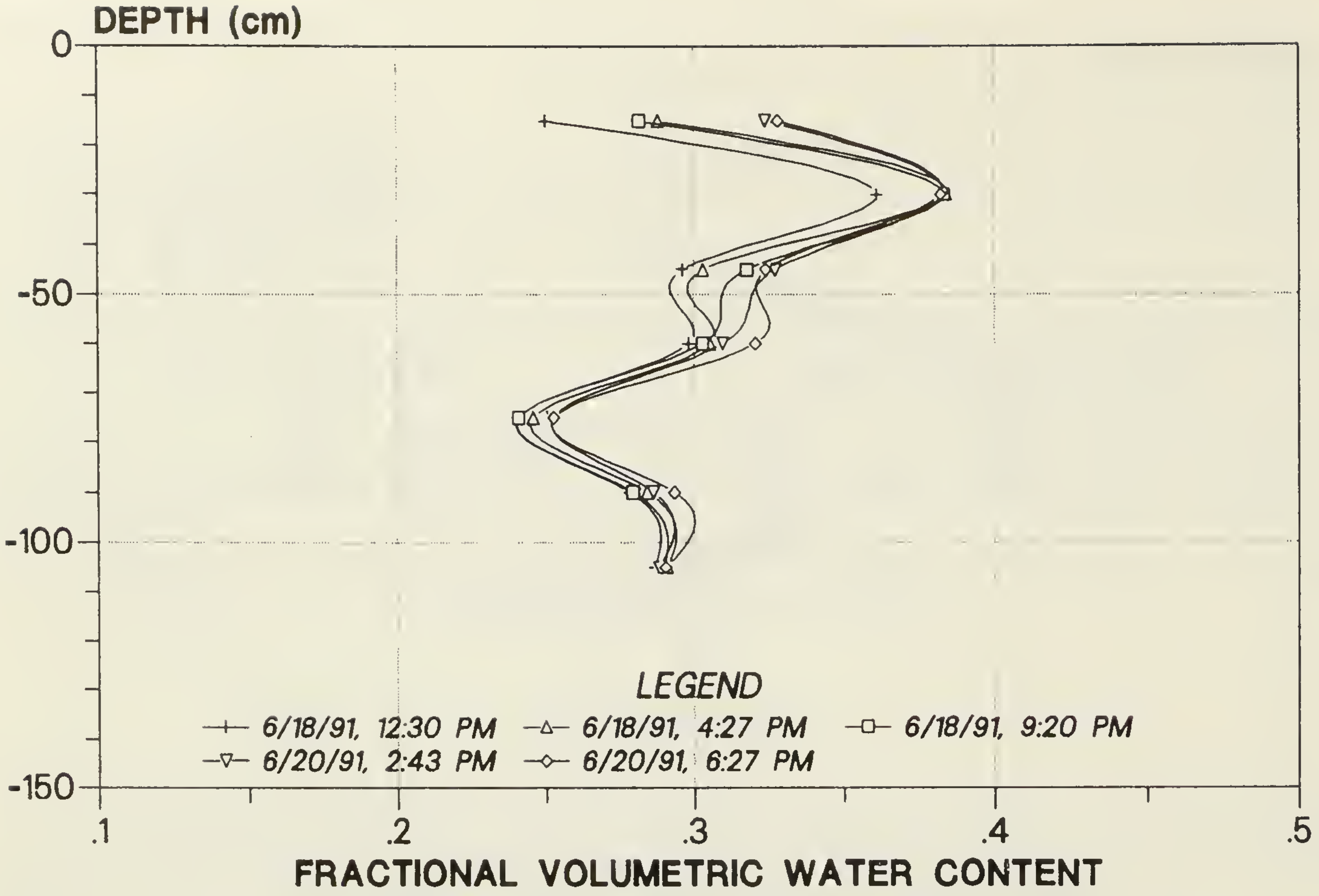


(a) Fresh water application, June 18, 1991 to June 20, 1991

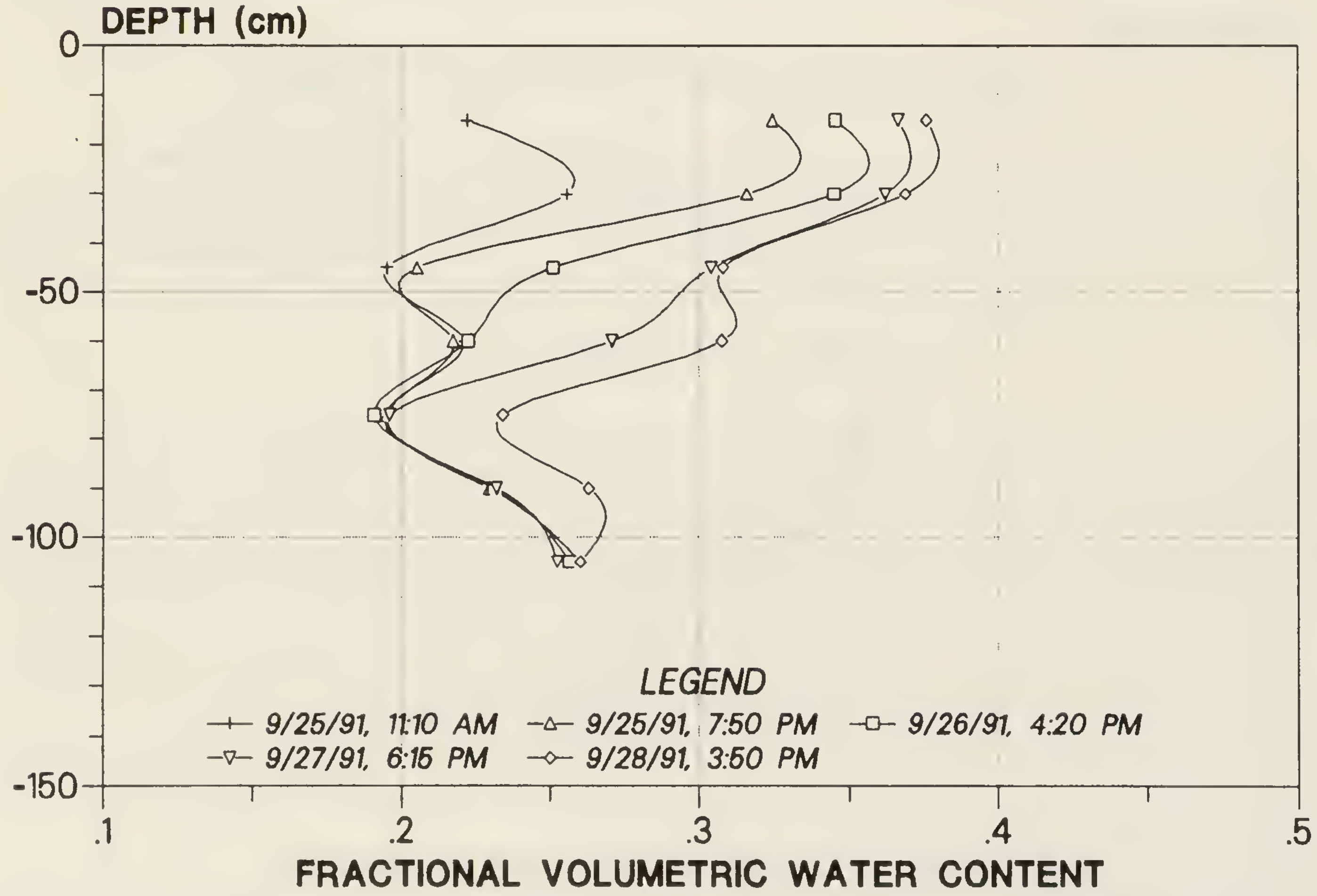


(b) Treated barren solution application, September 25, 1991 to September 28, 1991

Figure 1. **Soil moisture response at Site 1 during fresh water application (a) and during treated barren application (b).**



(a) Fresh water application, June 18, 1991 - June 20, 1991



(b) Treated barren solution application, September 25, 1991 - September 28, 1991

Figure 2. **Soil moisture response at Site 2 during fresh water application (a) and during treated barren application (b).**

Background Metals Concentrations

Pore water samples during the fresh water rinse indicated no detectable cyanide in the rinse water or soil complex. Soluble metal concentrations were consistently low, either below the detection limit or well below 1 mg/l, with the exception of manganese (Mn). The concentration of soluble Mn in pore water samples ranged from 0.25 to 4.87 mg/l. The occurrence of elevated Mn may be caused by changes in redox equilibria, exchange with other cations on clays, or exchange with other cations on organic chelating agents typical of organic-rich surface soils. Manganese concentration decreased slightly with continued fresh water application. This may be an indication that Mn was approaching some new equilibrium as determined by the characteristics of the applied water. Analysis of mine make-up-water taken in September, 1991 indicated no detectable Mn.

Treated Barren Application -- Metals Concentrations

Arsenic (As)

Peroxide oxidation treatment resulted in precipitation of As to a concentration of 0.007 mg/l in the applied treated barren, just above the limits of detection. Only two of the pore water samples contained detectable arsenic and then only at the same concentration as the treated barren solution. It appears that peroxide treatment is a highly effective method for arsenic removal.

Copper (Cu)

Because copper complexes well with cyanide in ore leaching processes, high concentrations are expected in mine process water. Precipitation of Cu by peroxide treatment during this study was about 90% efficient, reducing Cu concentration from 400 mg/l in the process solution to approximately 40 mg/l in the treated barren. Soil attenuation of soluble Cu was very effective at this LAD site. A trend of consistent attenuation with depth can be noted in Figure 3. Surface soils (0-16 inches) were at least 95% effective in Cu removal. After the applied treated barren had passed through 48 inches of soil, Cu removal was consistently greater than 99%. Copper concentration in pore water samples from the 48 inch depth ranged from 0.03 to 0.17 mg/l, well below the EPA Clean Water Standard of 1 mg/l. It is concluded that soils at this LAD site are very effective at attenuating Cu, even from initially high levels.

It should be noted that at closure, a more extensive treatment will be conducted to further reduce Cu levels. This treatment will include circulating barren solution through an ore circuit to promote cyanide destruction and Cu precipitation prior to peroxide treatment.

Iron (Fe)

Analytical results indicated a majority of samples had iron concentrations either below detection, or in the range of 0.04 mg/l. One exception was samples from the lysimeter on Site 2 at 30 inch depth which indicated Fe concentrations in the range of 1 mg/l. Because of the abundance of Fe in the soil environment, variability is not surprising. Since only one pore water sample exceeded the EPA Domestic Water Standard, Fe is not expected to be a significant problem in this system. The presence of soluble Fe may also be beneficial in the formation of insoluble ferrocyanide complexes.

Manganese (Mn)

Manganese appeared to be mobilized from the soil during the fresh water rinse. During the treated barren application, Mn was again released in concentrations higher than the solution being applied (Figure 4). Elevated Mn was detected at locations in the soil profile consistent with pore water analyses from the fresh water rinse. Metals which are effectively adsorbed by soils through exchange mechanisms (Cu and Ni for example), could displace exchangeable Mn. This hypothesis fits well with the observation that many metal ions were removed in the upper soil profile while Mn was solubilized.

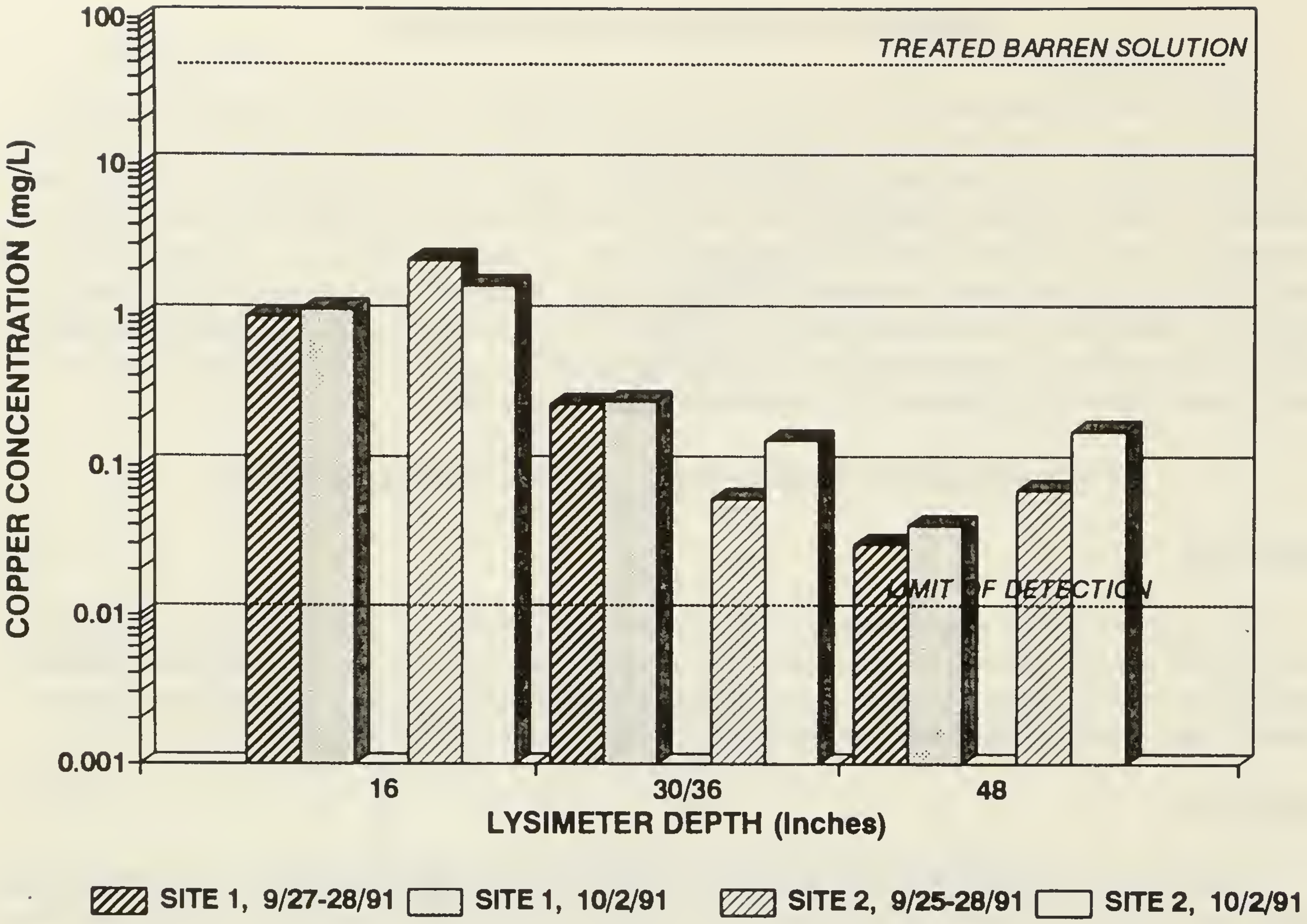


Figure 3. Copper in soil water samples during treated barren application.

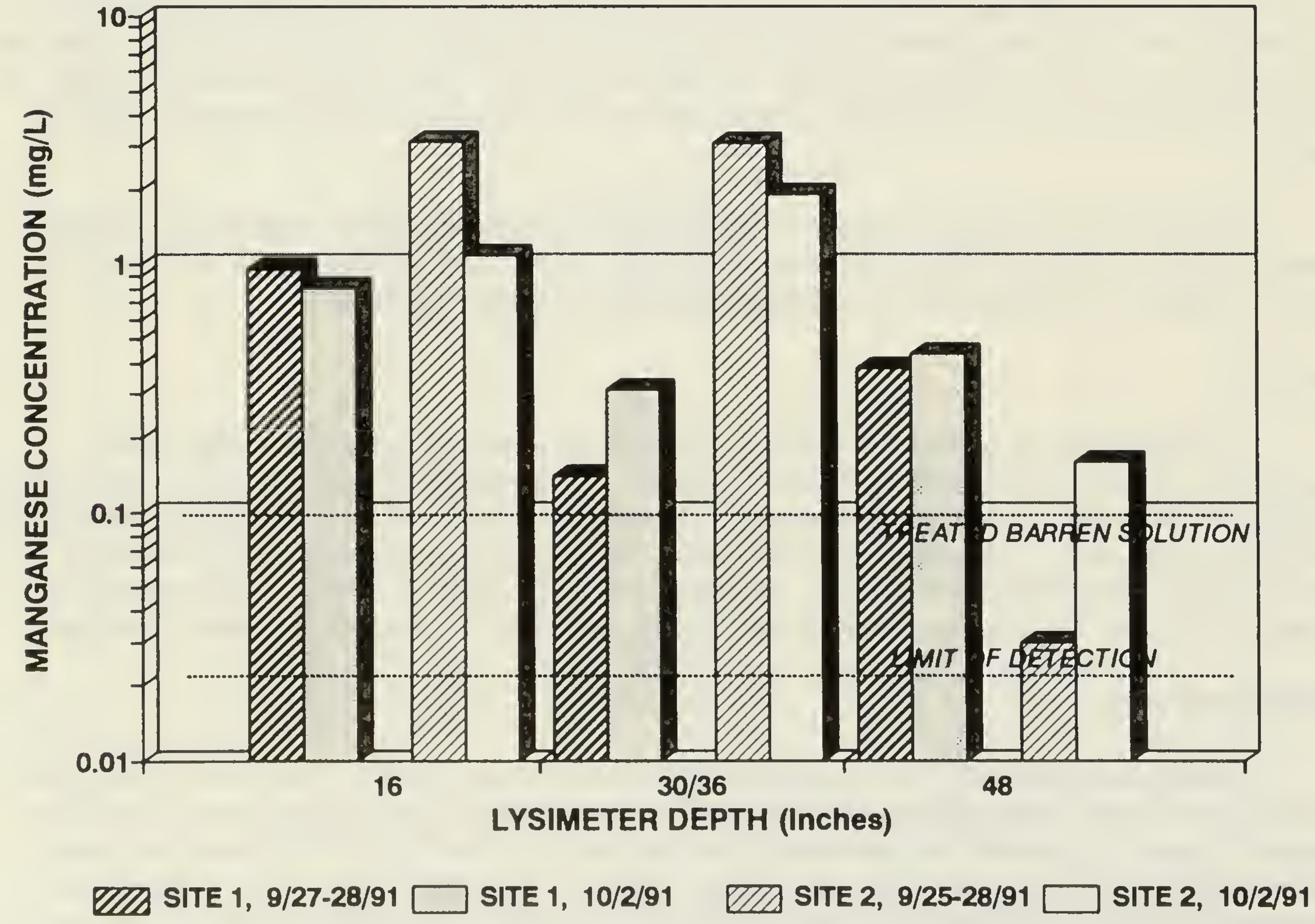


Figure 4. Manganese in soil water samples during treated barren application.

Manganese was detected in low concentrations in the treated barren and had declined to a similar level by the time solution had reached the 48 inch depth. This suggests that a majority of Mn released in the upper soil profile will be attenuated on clayey exchange sites with depth. At this LAD site, applied treated barren may pass through the soil profile with soluble Mn slightly exceeding the EPA Domestic Water Standard of 50 μg/l. However, Mn is an essential micro-nutrient and not particularly toxic at slightly elevated levels. EPA (1986) reported that Mn tolerance values for freshwater aquatic life range from 1.5 to 1000 mg/l and thus, manganese is not considered a problem in fresh waters.

In comparing data for manganese between the fresh water rinse (June, 1991) and the application of treated barren (September, 1991), pore water Mn concentrations at the 48 inch depth were virtually the same. In laboratory tests on soil samples from these sites using distilled water no manganese release was observed. These observations suggest that the release of manganese is dependent on the presence of ions normally present in many natural waters and is not the unique result of the application of process solutions.

Nickel (Ni)

Nickel concentration in the treated barren solution was 5.02 mg/l. The upper soil profile (0-16 inches) provided good initial attenuation and deeper soils (30-48 inches) reduced the Ni concentration to less than the MCL standard of 0.1 ppm in all cases (Figure 5).

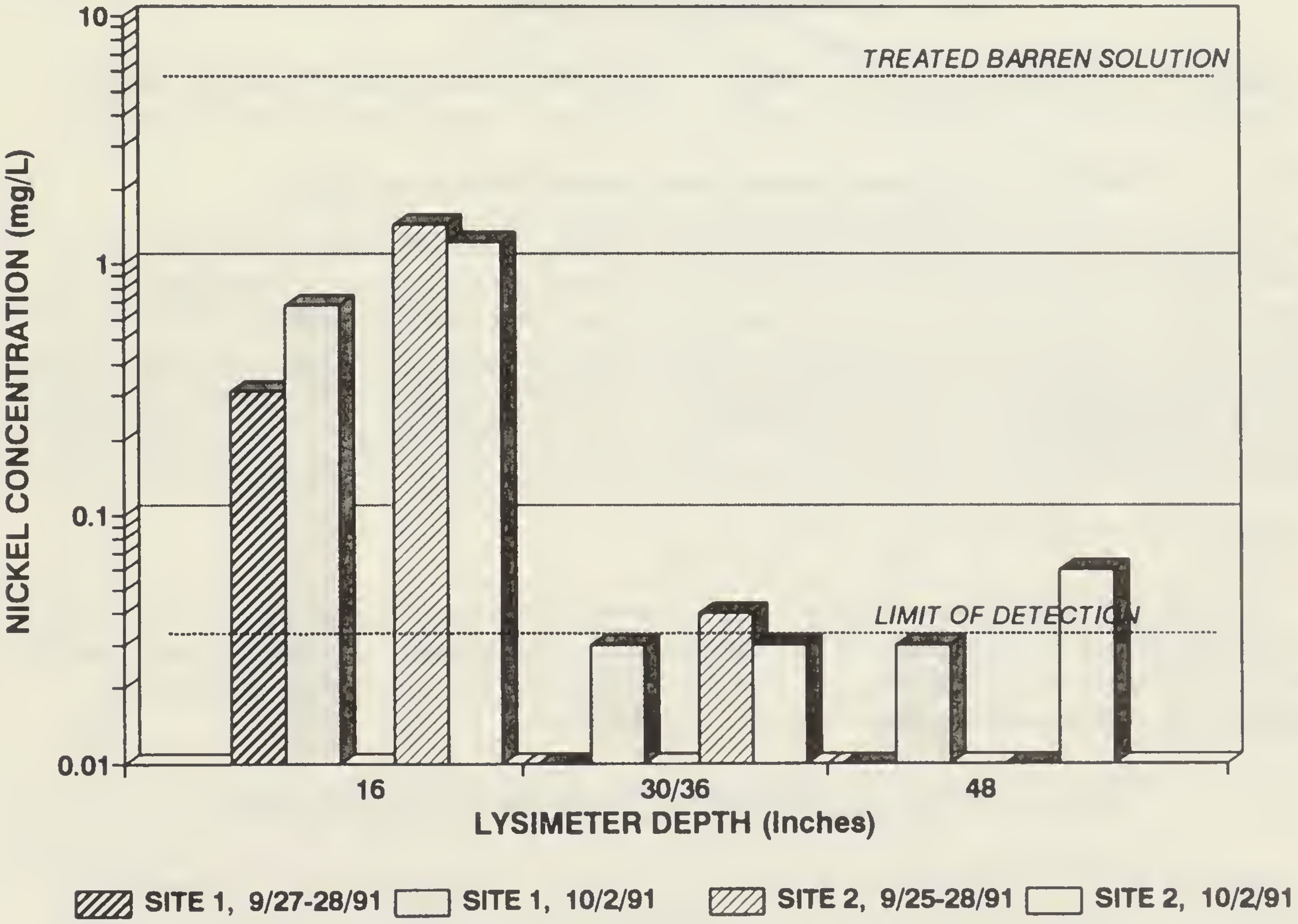


Figure 5. Nickel in soil water samples during treated barren application.

Selenium (Se)

Selenium was generally undetected in pore water samples during the fresh water rinse. Prior laboratory studies (Schafer and Hudson, 1990) found selenium was not effectively removed by peroxide treatment. In the same study, two inch soil columns in Tempe Cells achieved approximately 50% attenuation of Se applied in a treated barren solution. It was hoped that the deeper profile of a natural soil would result in improved removal, and this appears to be the case. Figure 6 indicates that

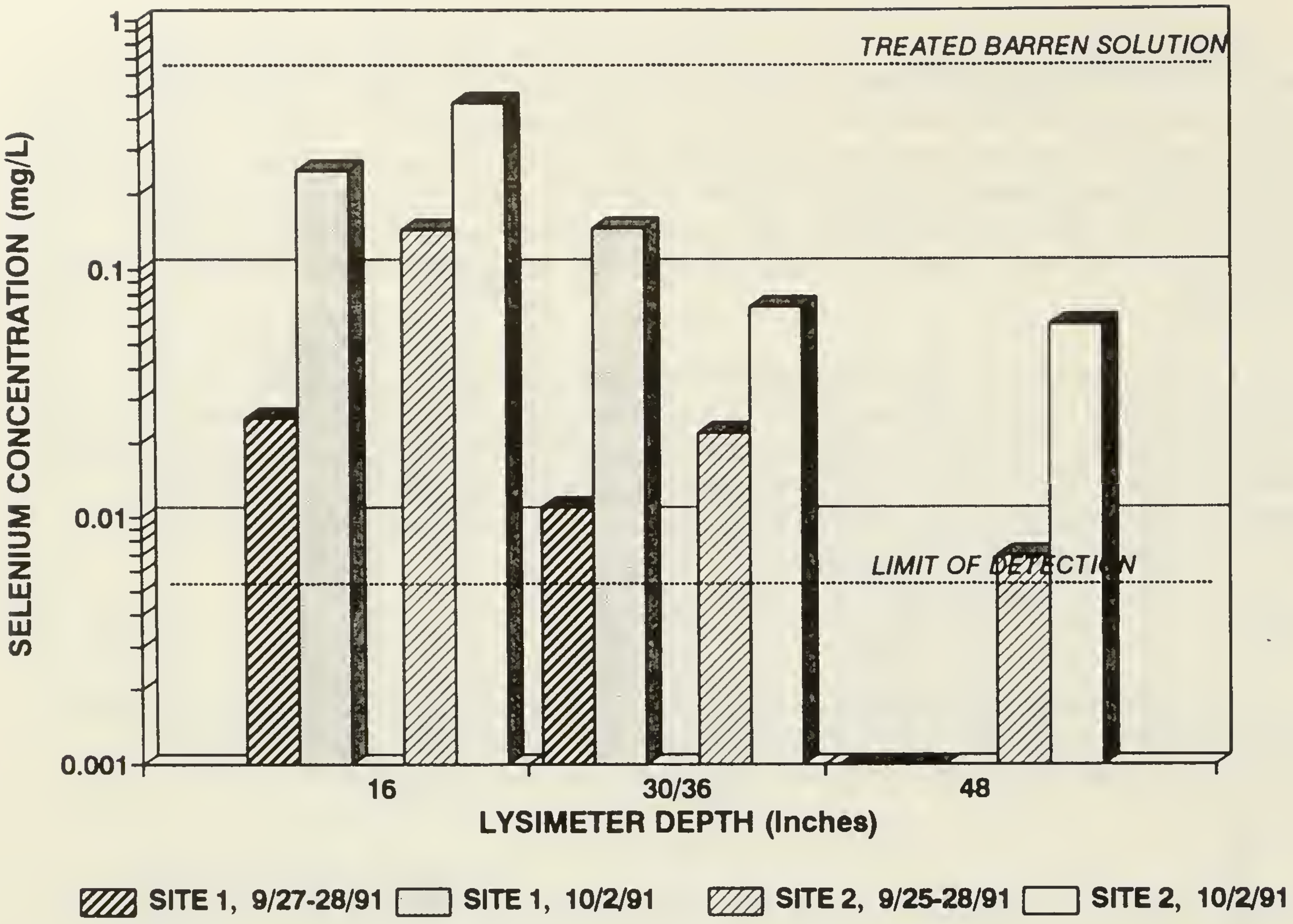


Figure 6. **Selenium in soil water samples during treated barren application.**

Se was generally removed by more than an order of magnitude when percolating through a deep natural soil profile. However, there is also a definite tendency for concentrations to increase in the "delayed" samples. This response was not apparent for most other monitored components. This suggests that the mechanism for Se removal is easily overloaded and is supported by the observations of Bar-Yosef and Meek (1987). They reported that Se adsorption is an important mechanism only at low selenium levels and that selenium is very mobile at high concentrations, especially in the selenate form.

Zinc (Zn)

Zinc present in the barren solution was reduced by peroxide treatment to a level of 0.14 mg/l. Upon land application of treated barren, an increase above the treatment concentration was noted in 16 inch depth pore water samples (Figure 7). However, soils deeper in the profile appear to be effective for attenuation of Zn released from the shallower soils.

Treated Barren Application -- Cyanide Concentrations

Cyanide is a widely used "lixiviant" in the gold industry. The reactive nature of cyanide contributes to its complex chemical behavior. Analytical techniques have been developed to analyze CN in its various chemical forms (Table 2). During the course of this study, an analytical problem was encountered. Because of difficulties obtaining adequate sample volumes for CN analysis (250-300 ml for manual distillation), an auto-distillation technique requiring only 25 ml was used on one set of total CN samples. Extraordinarily high total CN values on those samples raised concern. Research into the auto-distillation method (EPA 335-3) indicated that it was prone to interference from thiocyanate (SCN^-), a common cyanide degradation product which was found in the barren solution at a concentration of 600 mg/l. These suspect data were not used for interpretations in this report.

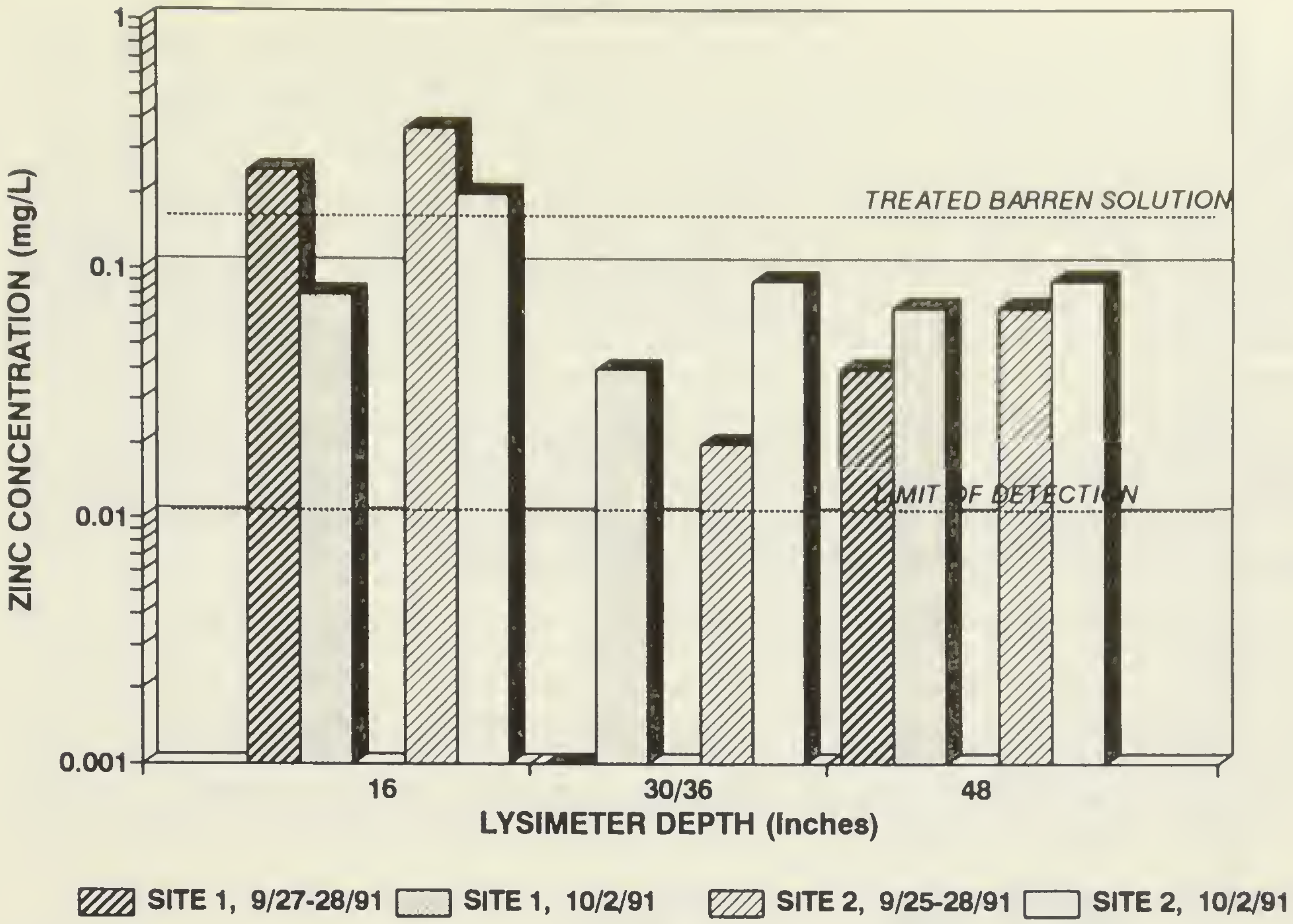


Figure 7. Zinc in soil water samples during treated barren application.

Table 2. Chemical forms of cyanide.

CYANIDE FORM	TYPICAL COMPONENTS	TOXICITY	ANALYTICAL RELIABILITY
Free Cyanide	CN^-, HCN, NaCN	High	Poor, many interferences
WAD Cyanide	$Cu(CN)_2^-$, $Cu(CN)_3^{-2}$, $Zn(CN)_4^{-2}$, $Ni(CN)^{-2}$ + free cyanide	Moderate[1]	Good, few interferences
Total Cyanide	$Fe(CN)_{6-4}$, $Fe(CN)_5^{-3}$, $Co(CN)_4^{-2}$ + WAD cyanide	Low[2]	Moderate, some interferences

[1] - Dissociation of WAD complexes at reduced pH releases free cyanide

[2] - Ferrocyanide while stable may partially dissociate free cyanide radicals under UV light

Free Cyanide

Peroxide treatment of barren process solution resulted in destruction of free cyanide to concentrations below the detection limit of 0.2 mg/l. No pore water samples or treated process solution samples collected during the project contained detectable free cyanide.

Weak Acid Dissociable (WAD) Cyanide

Cyanide in the WAD fraction was effectively destroyed by peroxide treatment. Initially, the barren solution WAD cyanide concentration was approximately 500 mg/l, and was reduced to 2.3 mg/l by treatment. In the soil complex, WAD cyanide was reduced by another one to two orders of magnitude. At the 48 inch depth, concentrations were below the Clean Water Standard of 0.2 mg/l (Figure 8). There appears to be a trend of continued reduction of WAD cyanide concentration with depth in the soil profile, particularly evident at Site 1. There is also a slight tendency for continued WAD cyanide reduction in the delayed samples indicating possible chemical or biological degradation. However, the fate of the WAD cyanide fraction is not clear. Total cyanide, which assayed 3.0 mg/l in treated barren, was not substantially reduced by contact with the soil. Since WAD cyanide, at 2.3 mg/l, is a substantial part of the total cyanide, one would conclude that the WAD fraction is converted to a strongly complexed form of cyanide. However, the concentrations of iron and cobalt which are known to form strong complexes, are too low to explain the total cyanide quantity reported. One possible explanation for this is that even the manual distillation technique for total cyanide may be subject to interference from the high levels of thiocyanate present producing high values for total cyanide when, in fact, most of the cyanide has been effectively attenuated or degraded in the soil.

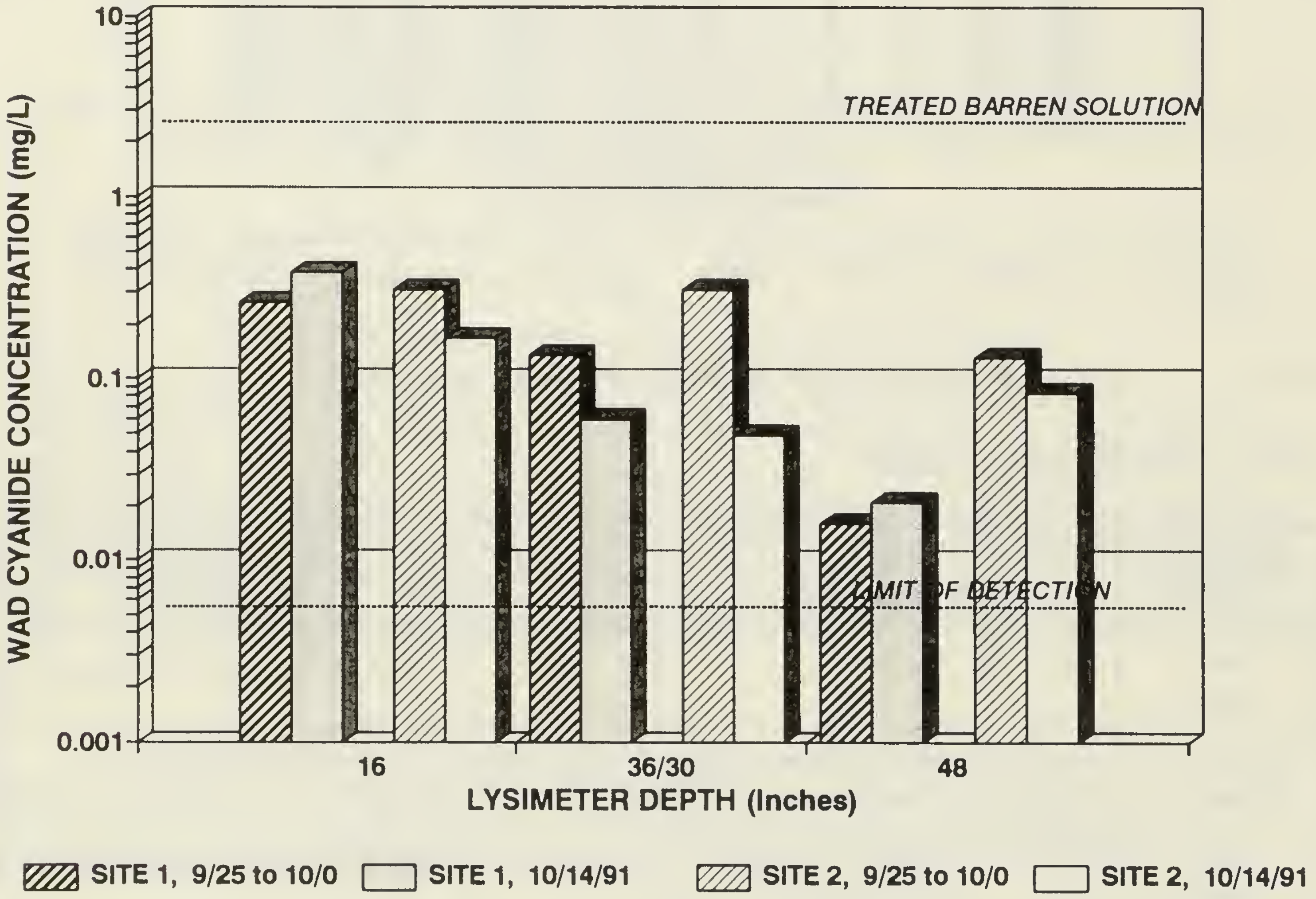


Figure 8. **Weak acid dissociable cyanide (WAD) in soil water samples during and after treated barren application.**

Total Cyanide

Peroxide treatment effectively reduced the total cyanide concentration in barren process solution from over 500 ppm to 3.0 ppm, representing greater than 99% efficiency. Pore water samples contained total cyanide concentrations ranging from 2.1 to 4.7 mg/l, which is similar to values in the treated barren solution. As previously discussed, the concentrations of strongly complexed metals such as Fe and Co cannot explain the high residual total cyanide concentration in the pore water samples in the absence of significant WAD and free cyanide fractions. Therefore, there is some suspicion regarding the validity of these assays.

CONCLUSIONS

On-site peroxide treatment in the field was effective for cyanide destruction. No free cyanide was detected in any of the samples collected. The majority of both WAD and total cyanide were also destroyed by on-site peroxide treatment. Removal of metals during peroxide treatment was less effective compared to previous laboratory tests because of a larger than expected change in pH during treatment. This need not be the case at time of closure. Lower levels of both cyanide and metals are anticipated in solutions receiving peroxide treatment at closure due to preliminary treatment by volatilization and photo-oxidation. The lower contaminant levels should produce less severe pH changes during peroxide treatment.

Residual metals were attenuated well by the soil complex. Copper was removed most effectively in the surface soil horizons possibly by organic exchange sites. Other metals, including zinc and nickel, appeared to be removed better in subsurface soils. Arsenic exhibited somewhat erratic behavior in the soil, probably because all analyses were near the limit of analytical detection. Manganese inherent to the soil was released by both treated barren application and by application of well water. However, manganese released from surface soils may be attenuated to some degree by subsurface soils. Peroxide treatment left significant amounts of selenium in the barren solution as it did in earlier laboratory tests. The soil complex was relatively effective in removing soluble Se. Pore water concentrations were 1 to 2 orders of magnitude less at the 48 inch depth than the applied barren solution. Selenium in pore water did display a tendency to increase in concentration with continued application. This trend was more apparent than with the other monitored soil water components, suggesting that the capacity of the soils for sustained selenium attenuation is limited.

Concentrations of WAD cyanide were significantly reduced by peroxide treatment, and further reduction in the soil complex was observed. When applied treated barren passed the 48 inch depth in the soil profile, all pore water samples were below the Clean Water Standard of 0.2 mg/l WAD cyanide. The pathways and interactions between WAD and total cyanide complexes were not clear. WAD cyanide concentrations declined while total cyanide did not, suggesting that the WAD fraction was being converted to strongly complexed metal cyanides. However, iron and cobalt concentrations were too low to explain the reported total cyanide concentrations. It is suspected that analytical interference may be producing unreliable total cyanide values.

Soil moisture measurements during application of the fresh water and the treated barren highlighted the highly variable nature of soils. Contained moisture varies significantly from season to season and this should be taken into account in a land application program. Also, the two test sites responded differently in their ability to accept water application. While both sites operated with unsaturated flow, Site 1 was eventually subject to subsurface flooding apparently because of low percolation rates encountered at depth. Such constraints limiting maximum application rates also need to be considered in managing a land application program.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

RESIDUAL CYANIDE DISTRIBUTION IN A NEUTRALIZED HEAP LEACH PAD

James Poell[1] and F.F. Munshower[2]

ABSTRACT

Kendall Mine Heap Leach Pad #1 is located in Fergus County northwest of Lewistown, Montana. The pad, constructed in the early 1980s, was most recently leached with sodium cyanide for gold extraction in 1988. After leaching, Pad #1 was neutralized in spring, 1989 with fresh water rinsing followed by alkaline chlorination using calcium hypochlorite. Thirty months later, the Reclamation Research Unit of Montana State University examined the pad. Samples were taken from 15 boreholes across the pad at incremental depths in order to determine the vertical and areal distribution and speciation of cyanide compounds. Three pore water samplers were installed to monitor cyanide and metal levels in the pad over spring and summer of 1992.

Analysis of solid ore samples shows a statistically-significant increase in total and weak acid dissociable cyanide levels at 10 - 17 feet depth. This may correspond to the interface of two lifts or benches of ore in the pad. Results from this investigation indicate that the pad has been effectively neutralized. However, without a standard cyanide analysis procedure for coarse, heterogenous ore samples or more intensive pore water sampling, definitive conclusions cannot be reached.

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[1] Graduate Student, Land Rehabilitation Program, Montana State University, Bozeman, Montana.

[2] Reclamation Research Unit, Montana State University, Bozeman, Montana.

INTRODUCTION

In the past 25 years there has been a wide expansion of the use of cyanide heap leaching by the mining industry. This process allows economic recovery of gold and silver from low-grade ore. Heap leach pads are constructed on slight drainage grades lined with an impervious clay or geotextile. Mined ore is then stacked and leached in successive ten to thirty foot lifts or benches on the pad. Sprinkler systems set on the surface of the lift to be leached deliver a high ph sodium cyanide solution that, in its percolation through the ore, forms stable complexes enhancing the dissolution of the metal. This "pregnant" solution is then collected as it drains from the pad for recovery and refinement.

After leaching, the heap leach pad still contains interstitial and adsorbed cyanide species, including free cyanide and metal-complexed cyanides. The metal-complexed cyanides are present in weak acid dissociable (WAD) metal complexes of cadmium, nickel, zinc and copper, and extremely stable iron and cobalt cyanide complexes (Smith and Mudder 1991). These cyanides, in sufficient concentration, have the potential to degrade surface and groundwater resources. If undisturbed, cyanide in the heap will degrade naturally. Natural degradation of cyanide can occur through volatilization, biodegradation, photodecomposition, and oxidation. Other degradation mechanisms associated with cyanide chemistry include complexation with transition metals, precipitation of complex cyanide compounds, formation of thiocyanate, and the hydrolysis of free cyanide (Smith and Mudder 1991). Engelhardt (1985) found that approximately 85 percent of the cyanide present in an untreated, inactive heap leach pad was degraded by natural processes over an 18 month period. How degradation will proceed, though, is site specific, depending on the interaction between geochemical conditions in the pad and site conditions such as precipitation, ore mineralogy, and ore permeability (Struhsacker and Smith 1990, Van Zyl 1988). To speed this degradation process, most mines use fresh or chemically-treated water rinses. Rinsing is continued until effluent draining from the pad meets required concentrations (0.22 mg/l WAD cyanide in Montana).

Despite acceptable effluent levels from passively abandoned or rinsed pads, later spikes or slugs of cyanide have been measured in conjunction with large precipitation events or further rinsing (Haight 1990, Struhsacker and Smith 1990). Residual cyanide in spent ore, pore spaces and/or blind-off zones diffuses out into this new wetting front as it passes, increasing its concentration and toxicity. Spent ore sampling that does not examine the full depth and breadth of a pad may not give a representative measure of neutralization success.

The concentration of cyanide throughout a decommissioned heap leach pad has not been thoroughly investigated. On behalf of the Hardrock Bureau of the Montana Department of State Lands and with the cooperation of Canyon Resources Corporation, the Reclamation Research Unit of Montana State University sampled a decommissioned heap leach pad in October/November of

1991. The objective was to determine the vertical and areal distribution and speciation of cyanide compounds within the pad.

STUDY AREA

The Kendall Mine is located in the North Moccasin Mountains, approximately 20 miles northwest of Lewistown, Montana. Gold has been produced in this area since the late 1800s advent of cyanide vat leaching. Open pit heap leach mining of gold began in 1982. After several ownership transfers, Canyon Resources in February of 1988 acquired Kendall Mine, including Heap Leach Pad #1, the focus of this investigation.

Pad #1 covers 2.4 acres and contains 90,000 tons of agglomerated ore on a compacted clay liner. The sandy gravel ore is comprised of approximately 62% gravel (including <1% stones up to 12"), 24% sand, and 14% clay. In August of 1988, the new owners ripped the pad to increase permeability and subjected it to cyanide leaching the next month. The pad was leached for 68 hours with over 200,000 gallons of sodium cyanide solution. No economically-recoverable gold was found and the pad was left to drain before spring decommissioning.

Active neutralization of Pad #1 began in March, 1989 with one week of fresh water rinsing. Prior to chemical neutralization, ore at the surface was sampled for cyanide. Results are shown in Table 1. Effluent discharge from the pad at this time measured 49.0 mg/l total cyanide and had a pH of 8.6. Neutralization of the pad with calcium hypochlorite followed, continuing through May, 1989. At this point, effluent draining the pad measured 0.1 mg/l total cyanide with a pH of 8.8. By the end of September, 1989, effluent draining the pad measured 0.02 mg/l total cyanide with a pH of 8.1.

Table 1. Cyanide levels in spent ore after fresh water rinsing at Kendall Mine Pad #1, March, 1989 (Haight 1990).

Location	Total Cyanide (mg/l)	WAD Cyanide (mg/l)	Free Cyanide (mg/l)
East Pad	<0.5	<0.5	<0.5
West Pad	2.1	1.2	<0.5

METHODS

Sampling Methods

Kendall Mine Pad #1 was sampled from October 29 to November 2, 1991, 30 months after active chemical neutralization ceased. Fifteen boreholes were located along two random transects across the pad (Figure 1). Samples were

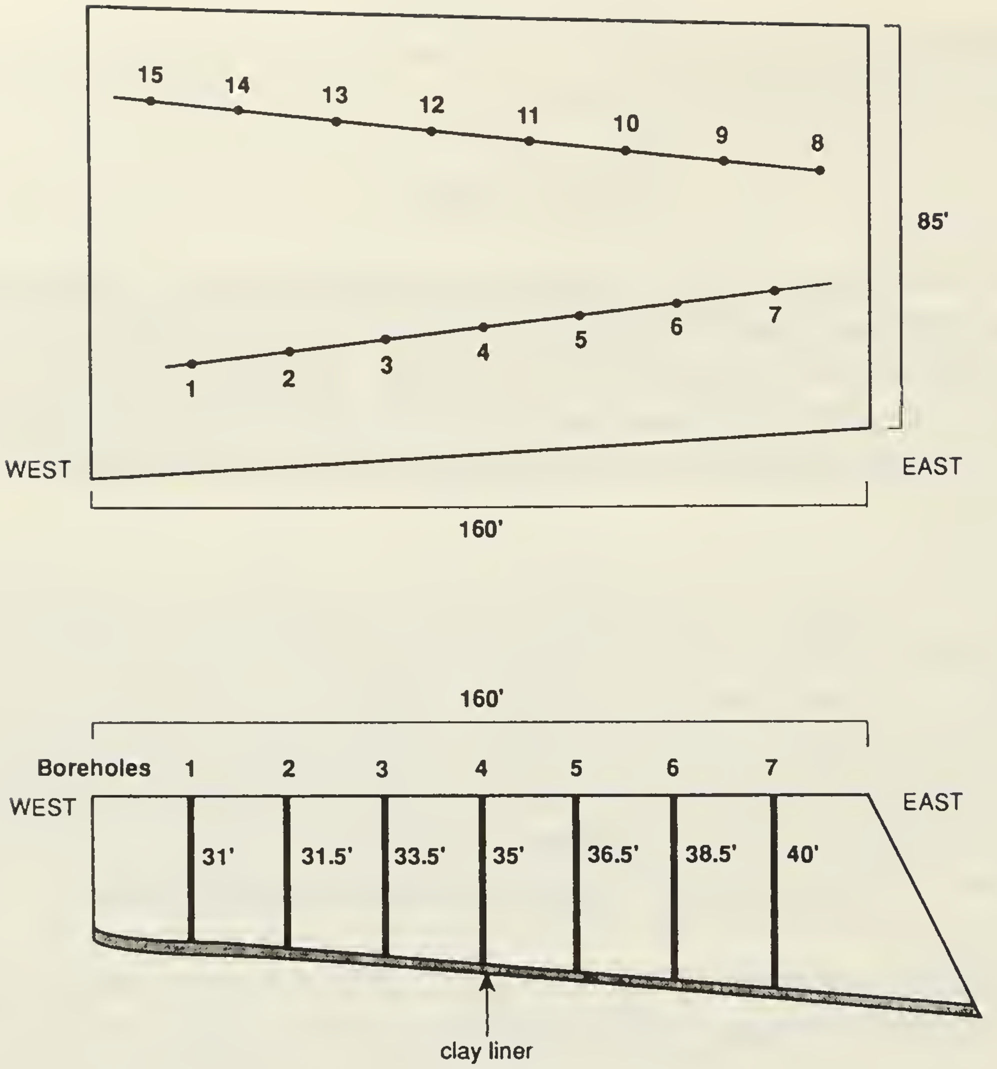

Figure 1. Location of sampling boreholes 1 - 15 across the surface (top diagram) and sampling boreholes 1 - 7 in cross-section (bottom diagram) through Kendall Mine Pad #1.

taken with a drill rig using a hollow stem auger and split spoon sampler pounded into the sampling zone. Samples were taken at ten depth increments (0-0.5, 0.5-1, 2-4, 5-7, 10-12, 15-17, 20-22, 25-27, 30-32 and 35-37 feet) down to the clay liner, approximately 40 feet deep. Split spoon samples were halved along the length of the spoon for composite cyanide and metals analyses, sealed in air-tight plastic bags, placed in coolers, and logged.

Split spoons were decontaminated between samples. Augers were decontaminated between boreholes. Any contact with the clay liner was backfilled with bentonite pellets. Boreholes were backfilled with drill cuttings after sampling was completed. Three boreholes were fitted with pore water samplers (suction lysimeters) to 26, 30, and 33 foot depths. These three suction lysimeters were used to monitor pore water quality in the unsaturated zone over spring and summer, 1992. Pore water samples were preserved to pH 12 using sodium hydroxide upon collection.

Analysis Methods

Because standard methods for a solids cyanide analysis applicable to mine ore were in a state of flux (Comba et al. 1992, Mudder 1991) analytical procedures for cyanide in liquids were modified to measure spent ore cyanide levels. Total and WAD cyanide were measured using ASTM (1992) Method D2036 reflux distillation of a solution containing one-half gram of <2 mm air-dried ore sample. Free cyanide in the ore solutions was measured using a cyanide ion-specific electrode (APHA Method 4500-CN-F). Hydrogen ion concentration was measured in pore water samples and saturated paste extracts of ore samples. Cadmium, copper, iron, nickel, and zinc levels in the solutions were measured using atomic absorption spectroscopy. Pore water samples were analyzed for total and WAD cyanide (ASTM Method D2306) and free cyanide (APHA Method 4500-CN-F).

RESULTS

Many spent ore sample analyses showed cyanide at or below detection limits. Further, due to the rocky nature of the ore, some samples contained insufficient in mass for analysis. Free cyanide, with two exceptions, was below detection limits. To clarify our interpretation, data from the ten depth increments sampled were combined (using depth-weighted averaging) into five depth categories. Those categories are 0-1.5, 2-7, 10-17, 20-27 and 30-37 feet. This data was then subjected to a weighted, two factor analysis of variance. Sample pH, total cyanide, WAD cyanide, and metal levels were compared as functions of depth category (Table 2) and borehole (Table 3).

Table 2. Mean levels[1] of cyanide, pH and water soluble metals in spent ore through Kendall Mine Pad #1.

Depth Category	Depth (feet)	Mean Total CN (µg/g)	Mean WAD CN (µg/g)	Mean pH	Mean Cadmium (µg/l)	Mean Copper (µg/l)	Mean Iron (µg/l)	Mean Nickel (µg/l)	Mean Zinc (µg/l)
1	0 - 1.5	1.3C[2]	0.5B[3]	7.9A[3]	9D[2]	195B[3]	291B[2]	196A[3]	124A[2]
2	2 - 7	1.5C	0.5B	7.7B	12BC	157A	328B	130A	138A
3	10 - 17	3.2A	1.1A	7.7B	10CD	145B	509A	162B	174A
4	20 - 27	2.4B	0.7A	7.7B	12B	80C	483A	116B	187A
5	30 - 37	1.5C	0.3C	7.8B	15A	60C	188C	35C	158A

[1] Mean values are based on N = 11, 12, 13, 14 or 15 observations.

[2] Multiple means comparison is based on LSD at 0.05 significance level using Student-Newman-Keuls Test (raw data). Means followed by the same letter in columns are not different.

[3] Multiple means comparison is based on LSD at 0.05 significance level using Student-Newman-Keuls Test (log transformed data). Means followed by the same letter in columns are not different.

Table 3. Mean levels[1] of cyanide, pH and water soluble metals for boreholes across Kendall Mine Pad #1.

Borehole	Mean Total CN (μg/g)	Mean WAD CN (μg/g)	Mean pH	Mean Cadmium (μg/l)	Mean Copper (μg/l)	Mean Iron (μg/l)	Mean Nickel (μg/l)	Mean Zinc (μg/l)
1	2.4ABC[2]	0.7BC[3]	8.2ABC[2]	17A[2]	113C[2]	447A[2]	94C[2]	135CDE[2]
2	1.3BCD	0.6BC	8.4A	11BC	133C	450A	89C	81E
3	3.2A	1.5A	8.3A	7C	318B	402A	421B	118CDE
4	3.3A	1.0A	8.2BCD	9C	738A	323A	705A	137CDE
5	1.5BCD	0.3BCD	7.6EF	7C	150C	245A	86C	219AB
6	1.3BCD	0.5BC	7.3G	7C	78C	250A	118C	180ABC
7	0.6D	0.2D	7.6EF	9C	57C	166A	28C	165ABC
8	0.7D	0.4BCD	7.4FG	9C	78C	235A	86C	227A
9	1.0CD	0.4BCD	7.9CD	6C	86C	277A	145C	81E
10	0.8D	0.4BCD	7.6EF	10C	83C	397A	57C	86DE
11	2.6AB	0.3BCD	7.4FG	12BC	76C	173A	49C	180ABC
12	1.6BCD	0.3CD	7.7DEF	11BC	146C	218A	69C	156BCD
13	1.4BCD	0.5BCD	7.8DE	15AB	107C	371A	116C	142CDE
14	1.7BCD	0.4BCD	7.9BCD	12BC	97C	589A	143C	117CDE
15	2.0BCD	0.6B	8.2AB	15AB	95C	418A	199C	92DE

[1] Mean values are based on N = 3, 4 or 5 observations.

[2] Multiple means comparison is based on LSD at 0.05 significance level using Student-Newman-Keuls Test (raw data). Means followed by the same letter in columns are not different.

[3] Multiple means comparison is based on LSD at 0.05 significance level using Student-Newman-Keuls Test (log transformed data). Means followed by the same letter in columns are not different.

Depth category 3 was significantly higher for total cyanide by depth. Depth categories 3 and 4 were significantly higher for WAD cyanide by depth. Mean borehole pH, total and WAD cyanide concentrations generally declined from upper to lower gradient (west to east) across the pad. Pad #1 pH is consistently between 7 and 8 through all depth categories, with no significant differences. Soluble metal concentrations show no distribution patterns or significant changes related to depth, borehole or cyanide/pH levels.

Pore water samples were collected every two weeks, March through September, 1992. Data are summarized in Table 4. One of the three suction lysimeters used to collect pore water was accidentally destroyed by a mine grader the winter

Table 4. Mean levels[1] of cyanide, pH and metals in pore water at Kendall Mine Pad #1, March - September, 1992.

1992 Sampling Date	Days Since Sampling Began	Mean pH	Mean Total CN (mg/l)	Mean WAD CN (mg/l)	Mean Cadmium (μg/l)	Mean Copper (μg/l)	Mean Iron (μg/l)	Mean Nickel (μg/l)	Mean Zinc (μg/l)
3/19	0	6.8	0.58	0.06	•[2]	•	•	•	•
4/10	22	6.7	0.88	0.06	50	58	703	141	442
4/25	37	6.6	0.55	0.09	26	24	396	76	259
5/07	49	7.4	0.55	0.10	23	29	775	86	247
5/20	62	7.5	0.66	0.10	23	48	208	76	145
6/06	77	7.6	0.67	0.11	26	19	498	87	172
6/21	92	7.2	0.72	0.10	30	15	577	76	326
7/09	110	7.0	0.76	0.11	30	24	313	76	436
7/24	125	7.6	0.81	0.13	33	10	333	65	140
8/09	141	6.4	0.83	0.31	30	0	350	76	240
8/23	155	7.9	0.49	0.09	26	5	556	54	256
9/06	169	7.6	1.20	0.19	•	•	•	•	•

[1] Mean values are based on N = 1 or 2 observations.

[2] Insufficient sample for analysis.

before sampling commenced. Examination of the data in Table 4, plotted as a function of time, show no increasing or decreasing trend in cyanide, pH or metal levels over the sampling period.

SUMMARY

Thirty months after active neutralization of Pad #1 with calcium hypochlorite, total and WAD cyanide concentrations peak at 10-17 feet depth (depth category 3). This is exhibited graphically in Figure 2. The peak may be attributable to textural changes at a lift interface. Coarser ore stacked on the second lift commonly rolls to the base, where it accumulates over the finer ore which comprises the top of the previous lift. Changes in flow paths at this interface or the development of blind-offs under the coarse material may have protected residual cyanide from neutralizing rinses.

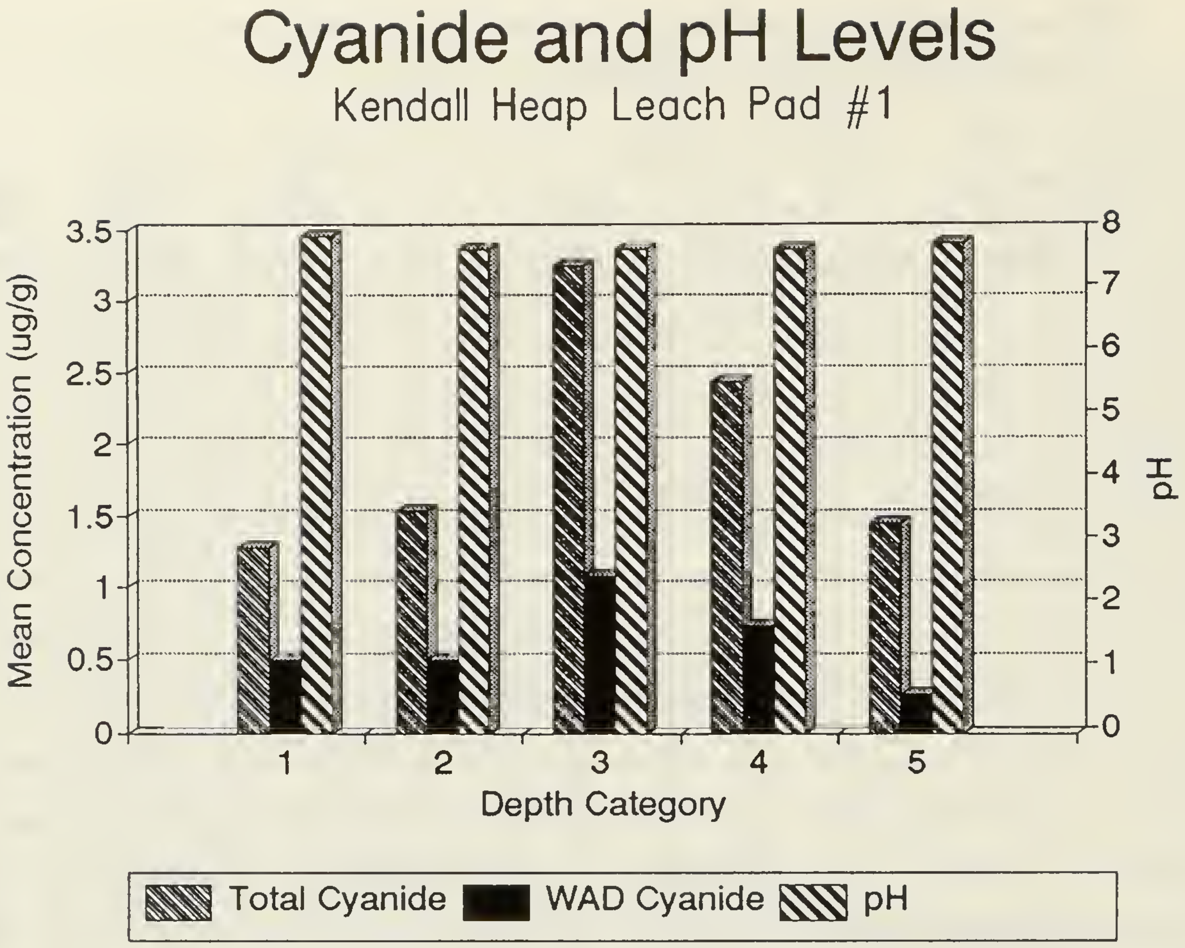

Figure 2. Mean levels of cyanide and pH in spent ore through Kendall Mine Pad #1.

Results from this investigation would indicate that Pad #1 has been effectively neutralized. However, a definitive conclusion regarding the neutralization of cyanide within this pad cannot be made without recognition of the difficulty in measuring cyanide concentrations in heterogenous spent ore. Smith and Mudder (1991) and preliminary results from ongoing Bureau of Mines research (Comba et al. 1992) indicate that air drying ore samples prior to analysis can result in a loss of cyanide. Techniques in preservation/processing that reduce cyanide loss from coarse, heterogenous ore samples, however, suffer from high variability and poor precision (Comba et al. 1992). Further investigation of the correlation between cyanides and soluble metals in pad effluent (accurately measurable and more easily obtained) and cyanides and soluble metals in pad ore would be helpful to both industry and regulators in decommissioning efforts.

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Planning, Rehabilitation and Treatment of Disturbed Lands
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Bird Mortalities Associated With Mining and Oilfield Waste Pits in Wyoming

Pedro Ramirez, Jr.[1] and Brent J. Esmoil[2]

ABSTRACT

Trona mine and oilfield waste pits are causing migratory bird mortalities in Wyoming. The number of migratory birds killed in these waste pits in Wyoming is unknown; however, estimates for other areas are available. An estimated 100,000 waterfowl are killed each year in oil pits throughout Oklahoma, New Mexico, Kansas and the Texas panhandle; and an estimated 1.5 million waterfowl are lost each year to oil pits throughout the United States. Birds are attracted to mining and oilfield waste pits by mistaking them for natural bodies of water. Birds entering these pits become entrapped in the waste and die from exposure or from ingesting oil or other hazardous substances.

Field observations and studies have demonstrated that deterrents such as flagging, strobe lights, metal reflectors and noise makers are not effective at preventing bird mortalities from occuring in these pits. Netting appears to be the most effective technique to prevent birds from entering waste pits. The U.S. Fish and Wildlife Service conducted seminars in Nebraska, Montana and Wyoming to inform the mining and oil industries of the problem and alternatives available to prevent bird mortalities in waste pits.

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[1] U.S. Fish & Wildlife Service, 2617 E. Lincolnway - A, Cheyenne, WY 82001

[2] U.S. Fish & Wildlife Service, 203 W. Second St., Grand Island, NE 68801

INTRODUCTION

Impoundments ranging in size from small stock ponds to large reservoirs provide vital stopping sites for migratory aquatic birds in Wyoming. Aquatic birds have trouble distinguishing pristine wetlands from small ponds and reservoirs containing oil or other hazardous wastes. In Wyoming, oilfield waste pits and other industrial wastewater ponds create a fatal attraction for waterfowl, songbirds and shorebirds. Oil and mining wastewater ponds are causing the deaths of over 1.5 million migratory birds each year in the western United States. Many waste pits occur in oilfields throughout Wyoming and other oil-producing states. Industry uses the pits to contain water discharged from oil wells. Ineffective separation of oil from the water results in an oil-covered pond creating a death trap for migratory birds. Birds mistake the oily pits for water in the semi-arid west.

DESCRIPTION OF THE PROBLEM

Wildlife mortality in oilfield waste pits has been documented in several states. During 1951 and 1952 King (1956) surveyed oilfield waste pits in Wyoming and found 914 waterfowl carcasses in 65 pits. Tully and Boulter (1970) documented 585 vertebrate carcasses in 41 oilfield waste pits in Colorado. In the San Joaquin Valley of California, Thomas (1971) reported over 1600 birds and mammals in waste pits. Flickinger (1981) reported 476 vertebrates in 61 oilfield waste pits in the Texas gulf coast. In Wyoming, Esmoil (1991) reported a total of 282 vertebrate carcasses in a survey of 35 oilfield waste pits in 1989 and 334 carcasses in 53 waste pits in 1990. The total mortality recorded (616) consisted of 41% passerine birds (Order Passeriformes), 33% mammals, 14% ducks (Family Anatidae), 5% shorebirds (Order Charadriiformes), 5% shrikes (Lanius ludovicianus) and 2% raptors (Order Falconiformes).

Birds landing in oil waste pits can suffer death in several ways. They can become entrapped in the oil and drown. Birds can ingest toxic quantities of oil by preening their oil-covered feathers. Birds can also die of cold stress if the oil damages the insulation provided by the plumage. Breeding birds using oil pits can transfer oil from their feathers to their eggs. Microliter amounts of oil applied externally to egg shells can cause embryo mortality (Albers 1977, Hoffman 1978, King and Lefever 1979). Waterfowl ingesting sub-lethal amounts of oil can produce eggs with altered yolk structures which reduces hatching success (Grau et al. 1977).

Oil waste pits are not selective in the species of birds they attract and kill. Esmoil (1991) found waterfowl, shorebirds, songbirds and raptors were fatally attracted to oilfield waste pits in the Bighorn Basin of Wyoming. Esmoil (1991) also found traditional deterrents such as colored flagging, metal reflectors and strobe lights were ineffective in deterring birds from waste pits. Esmoil (1991) did not find any bird mortality in pits covered with netting. Netting appears effective at keeping birds from entering the waste pits. Netting was also found to be the most effective technique in preventing bird mortalities in oilfield waste pits in the southwestern United States (Jim Hubert,U.S. Fish and Wildlife Service, Albuquerque, New Mexico, personal communications, July 1991).

Wastewater ponds at trona (soda ash) mines and some coal-fired power plants also kill birds. Trona is mined in the Green River Basin in Wyoming and is used as a base for detergents and a variety of other products. Trona evaporation ponds contain high concentrations of sodium decahydrate that crystallizes at temperatures as high as 70° F. Birds landing on the ponds will experience crystallization of this compound on their feathers. The crystallization can destroy the insulative qualities of the feathers causing the bird to die of exposure. The crystals also can weigh the bird down, causing it to drown. Birds also can ingest the crystals during preening and die of sodium toxicity.

Because of their high alkalinity, trona evaporation ponds remain ice-free longer than nearby freshwater ponds, rivers and lakes. During the cold season when all other waterbodies are frozen, aquatic birds migrating through the area will flock to the open water at the trona ponds. The risk to birds is greatest at this time as crystallization and hypothermia are enhanced by the colder temperatures.

Some coal-fired power plants use trona wastewater to neutralize the acidity of their smokestack scrubber water. Water is used to remove sulfur dioxide from smokestack emissions. The scrubber wastewater is discharged into evaporation ponds. Hazards to birds in these ponds are similar to those at the trona evaporation ponds. Bird mortalities have been documented at the Jim Bridger Power Plant at Point of Rocks and at the Naughton Power Plant near Kemmerer, Wyoming (Jim Klett, Special Agent, U.S. Fish & Wildlife Service, Lander, Wyoming, personal communications, 1992).

Wastewater ponds at heap leach gold mining operations also kill migratory birds. These mines usually contain ponds with toxic concentrations of cyanide and metals that can kill birds on contact. Cyanide is used in the heap leaching process to separate small amounts of gold from the ore. The gold ore is piled in a lined pad and a cyanide solution is sprayed on the crushed rock to separate gold from the low-grade ore. Currently, there are no heap leach gold mining operations in Wyoming; however, heap leach gold mining is done in the neighboring states.

WHAT'S BEING DONE ABOUT THE PROBLEM?

The U.S. Fish and Wildlife Service has informed the oil and mining industries that bird mortalities caused by waste pits are a violation of the Migratory Bird Treaty Act (MBTA). The U.S. Fish and Wildlife Service is mandated to enforce the MBTA, first enacted by Congress in 1918 in a treaty with Great Britain to protect migratory birds. Companies can be penalized up to six months in jail and a $10,000 fine under the Migratory Bird Treaty Act.

State and federal regulatory agencies are taking steps to prevent bird mortalities from occurring in oilfield waste pits. The New Mexico Oil and Gas Commission has ordered oil operators to cover waste pits with netting to protect migratory birds (Jim Hubert,U.S. Fish and Wildlife Service , Albuquerque, New Mexico, personal communications, July 1991). The Wyoming Oil and Gas Commission has enacted regulations requiring the oil industry to cover oilfield waste pits with netting if they pose a hazard to migratory birds.

Federal agencies regulating oilfield operations also can require preventive measures such as netting through their permitting procedures. The U.S. Fish and Wildlife Service has requested the Bureau of Land Management's Wyoming State Office in Cheyenne to consider revision of their oil operations permit requirements to protect migratory birds.

The U.S. Fish and Wildlife Service has conducted seminars in the southwestern United States, Nebraska, South Dakota, Montana and Wyoming to inform the oil and mining industries of the bird mortality problem and available alternatives to prevent migratory bird deaths. The seminars have been effective. Oil companies have shown their willingness to correct the problem in New Mexico, Texas, Nebraska, Kansas and Wyoming. Several oil companies have covered their oil waste pits with netting while others have removed the pits altogether. In the Bighorn Basin of Wyoming, Bass Enterprises eliminated almost 30 percent of their waste pits at the Cottonwood Creek Oilfield (Ken Moore, Production Foreman, Bass Enterprises, Wyoming, personal communications, August 1991). Bass Enterprises covered the remaining 71 waste pits with netting to prevent bird mortalities.

Individuals or companies that fail to take preventative actions may find themselves at odds with the U.S. Fish and Wildlife Service and the U.S. Attorney's Office. The U.S. Attorney's Office in Wyoming has promised to vigorously prosecute illegal killing of migratory birds. Several court cases have established a legal precedence for the taking of migratory birds by wastewater ponds (United States vs. FMC Corp., United States vs. Corbin Farm Service).

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Planning, Rehabilitation and Treatment of Disturbed Lands
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SMCRA-INCEPTION, MATURATION, AND WHAT?

L.L. Byrd[1]

ABSTRACT

The Surface Mining Control and Reclamation Act (SMCRA), which was enacted in 1977, has matured from a simple and misinterpreted piece of legislation to a very sophisticated and effective program. The early development of permitting and enforcement programs in the respective states throughout the country has been described as eventful, chaotic, questionable, desperate, a drain to the well being of the coal industry to a "miracle' for the naive environmentalists throughout the country. This paper will describe the various stages associated with the growth and development of regulatory programs of the western states with particular interest given to the State of New Mexico's program development, which has evolved to an experienced program.

The early development of the regulatory programs involved the drafting of rules and regulations which could be enforced in the field and in turn hold up in a court of law. The regulations of any environmental type of legislation are the key to ensuring that the environment is protected along with ensuring that the industrial base of this country is not inhibited or shut down by biased and subjective rules which favor industry or the environment. Once regulations were in place, they were enforced out in the field, and this is where the adequacy and validity of the respective rules was tested. The early inspectors were faced with the challenge of enforcing the early rules and expecting industry to abide by the law. Informal hearings, formal hearings, commission hearings and court cases during the early days of SMCRA were sometimes very lengthy and filled with emotional outbursts caused by inappropriate regulations, defiant and aggressive operators and above all serious growth pains associated with the implementation of legislation which affects industrial development of an industry which had developed on its own for over one hundred years.

The present condition of SMCRA at this time is stable and functional. The operators are mining and reclaiming according to the law, and this is a good sign that the regulators and industry have adapted very well to the legislation and rules.

The future of SMCRA will involve bond release of lands reclaimed which were seeded at a minimum ten years ago. This will be a very delicate process which will involve scientists, administrators, technicians and politicians. It is very important that the right people be involved in the compromise which will be required of the respective vegetative standards for the purpose of releasing the reclaimed lands so that post mining land use can commence. This paper will discuss the direction that SMCRA is heading, and will also recommend appropriate actions to insure that the lands being released will be managed for post mining land use.

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[1] Mining and Minerals Division, 2040 South Pacheco St., Santa Fe, New Mexico.

SMCRA-INCEPTION, MATURATION, AND WHAT?

The Surface Mining Control and Reclamation Act (SMCRA) was enacted in 1977. The primary purpose of the legislation was to protect the environment and health of citizens and ensure that post mining land use is restored to areas affected by the mining operations. The initial response to SMCRA was diversified, damned or blessed depending on what entity one was associated with. The coal mining industry reacted to the legislation as being temporary in nature, and with wishful thinking that the industrial giants in this country would prevail in performing minimal reclamation activities.

Typical behavior of industry included the following: (1) The submission of substandard permitting documents including baseline data; (2) The continual confrontational attitude toward state and federal inspectors conducting environmental protection inspections on their respective mine sites; (3) Appealing every Notice of Violation written; (4) Continuing to mine without much regard for the regulations; (5) Placing the State Regulatory Authorities in between the operator and the Office of Surface Mining in order to confuse and complicate the ultimate goal of performing effective reclamation practices by restoring the land to a condition as good as or better than when the mining operation commenced and (6) Conducting normal operations with all apathetic attitude towards meeting the standards and regulations of SMCRA.

The mining industry was in a tenuous position since personnel did not understand the far-reaching ramifications associated with SMCRA. The main priority for coal mines was PRODUCTION with the mission of protecting the environment being last in the terms of goals and budget constraints. Generally, industry Environmental Coordinators lacked the authority to pull off equipment and resources needed for reclamation and environmental protection from production type operations. This made it very difficult to conform to SMCRA, even in times when the personnel representing environmental quality had the right intentions of following the pertinent environmental laws.

Management personnel representing the coal industry were faced with the choice of attempting to follow SMCRA, or to "get by", by performing minimal environmental assignments in a haphazard or apathetic manner. Upper management has to be behind any type of decision which affects daily operations, and especially anything as sensitive as preforming tasks and operations to protect the environment, which in turn cuts into the maximization of profit of the mine.

The agencies responsible for permitting and enforcing SMCRA were initially faced with the task of enforcing an Act and regulations which were viewed by industry as an "obstacle" and a "waste" of capital to protect the environment. Early inspections were confrontational, and personnel conducting the inspections did not know how far their authority stretched, nor did they always know if their supervisor, usually a political appointee, would support their actions. How could a government agency shut down a mine, and should they have the authority to do so? The coal operators challenged inspectors by intimidation, political pressure, apathetic attitudes, delay tactics on the mine as well as the submittal of substandard required information and also by showing an attitude of complete refusal to cooperate with the idea of some "naive" unqualified personnel being able to influence the activities on a mine site.

In some cases the operator was right in accusing the regulatory authority of employing personnel without mining experience or wide-ranging backgrounds which would indeed force industry into making decisions that would be governed by an incompetent and untrained regulatory authority. Most states did not have effective training programs to indoctrinate new employees to perform objective and truly comprehensive inspections. This was unfair to the inspectors and personnel attempting to regulate and perform their duties in a professional manner without the proper tools to effectively carry out the required regulatory function. This of course brought out cries from industry that the coal industry was being forced to do things without proper guidance. In some instances violations were written without proper mitigative measures. One of the most important componets of a Notice of Violation is the required mitigative action. Improper mitigation measures can be very expensive, and in some cases the environmental degradation has continued, or was accelerated due to poor mitigation plans and actions.

At this time the coal mining industry and the Regulatory Authorities are cooperating and working together towards the unified goal of protecting the environment and the health of the citizens. Permits are being submitted and issued with good baseline data as a means to insure that environmental protection is being practiced along with the goal of maximizing coal production. This is where the SMCRA has evolved in the State of New Mexico, and hopefully this is how respective programs are being conducted nationwide. Permit modifications revisions, and deviations from the original mine plan are being submitted with enough lead time for the regulatory authority to comprehensively review to ensure environmental compliance. This is a step forward, and I believe that industry has adapted very well to conforming to the respective regulations. Most companies are at least as concerned with environmental protection as the regulatory agency.

Quarterly and monthly inspections have progressed to the point where it is quite common to inspect a mine and not issue a Notice of Violation. This is a step forward and industry should be commended for a job well done.

Oversite inspections conducted by the Office of Surface Mining have helped the respective states enforce the Act and regulations, however, it should be pointed out that the respective states indeed have primacy. The interpretation of the Act and the regulations is the responsibility of the regulatory authority of the respective states who have primacy. This is one subject in which the Office of Surface Mining and the states have to come to compromise in order that the Act and regulations can be enforced in a fair and objective manner with the state being the primary regulator. Standardized oversite inspections throughout the country would also be a step forward for the objective implementation of SMCRA nationwide. This would also allow industry to function with a degree of confidence in knowing that the respective states that have primacy have the last say in the enforcement of their respective programs.

Bond release is really the final formal phase in the implementation of SMCRA, and many states are and will be in phase III bond release mode for years to come. The phase III bond release is important in all aspects including stability of the landscape, vegetation production, vegetation diversity and also ensuring that the prescribed post mining land use is achieved. Vegetation success standards are critical in the release of SMCRA lands, and it is important to point out that is some cases the standards will not be met. This is a critical area, and I believe that a compromise will have to be negotiated in order that the lands can be released. The compromise should be worked through by competent vegetation specialists, soil scientists, the landowner and in some cases administrators, HOWEVER, it is critical that the issue not be solved politically.

Careful and logical decisions need to be made with realistic and accurate data being the key to making the right choice when it comes to deviating from the prescribed vegetative success standards which were set with good intentions early on in the permitting process.

The management of lands after bond release is critical, and it should be noted that the lands need to be managed consistent with criteria furnished by Federal and State agencies who have the experience and expertise necessary to ensure that the released lands will not fail by being overgrazed or not managed properly. It is necessary that the landowners and the respective agencies that oversee land management come together and implement sound management principles to maximize the effectiveness of reclaimed lands that have been released back to the original landowner. The proper management of post released SMCRA lands is as critical as submitting and implementing an effective reclamation plan during the early permitting stages of a mine, and it is important to note that PROPER MANAGEMENT is the key to completing the intent of SMCRA. I am confident that the respective states and OSM will strive toward the goal of ensuring that effective land use management is an integral part of their existence, and that all personnel will strive to perform their duties and responsibilities with that goal in mind.

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INTEGRATED SUITABILITY CRITERIA FOR SOIL AND OVERBURDEN MATERIALS

S.E. Fisher, Jr.[1] and T.H. Brown[2]

ABSTRACT

The importance of detailed planning for mine soils being constructed in drastically disturbed land reclamation can not be overstated. Much of the productivity and stability of the reclaimed site is dependent upon the quality of reconstruction of mine soils which have the capability of supporting the immediate and long-term land uses set forth in the reclamation plan. As is the case in natural ecosystems, the characteristics of the soil resource (mine soils in disturbed/reclaimed environments) are one of the primary determinants of site productivity and stability. However, it must be understood that site characterization is an integrated evaluation of interrelated parameters that determine the final productivity of the reclaimed site. Therefore, criteria for reclamation success can not be dependent on single parameters, but must be based on an integrated evaluation of interrelated considerations.

Reclamation planning currently emphasizes short-term accomplishment of reclamation goals to meet the requirement for determination of successful reclamation and bond release. In an era of austerity, such as we are currently experiencing, the reclamation planner must accomplish the most effective biological/resource based reclamation in the most monetarily efficient manner. To achieve these goals of effective and efficient reclamation, it is necessary to develop an understanding of the impact of pedogenic processes on disturbed landscapes and undertake long-term projection or modeling of these changes. Society can no longer bear the costs of reclamation failures, short or long term, and the enormously increased costs associated with attempts to mitigate or rehabilitate such sites.

This paper will discuss the development of soil and overburden suitability guidelines based on multiple types of analyses. For illustration, sodicity and acid-forming materials problems will be discussed. However, other problems that are important to successful reclamation of mined lands will also require a multivariable approach. Problems associated with elements such as B, Se, and many metals will require more elaborate criteria. The other important aspect of this paper will deal with the depositional or surficial environment where a soil or overburden is placed. The pedogenic and geochemistry processes, as well as other weathering processes, will significantly influence the suitability of the soil and overburden materials for reclamation purposes.

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[1] Office of Surface Mining, Brooks Towers, 1020 15th Street, Denver, Colorado.
[2] Western Research Institute, P.O. Box 3395, Laramie, Wyoming

INFLUENCE OF RHIZOSPHERE MICROORGANISMS ON UPTAKE OF SELENIUM BY TWO-GROOVED AND TINE-LEAFED MILKVETCHES

Stephen E. Williams[1]

ABSTRACT

In the 1940's, O. A. Beath and others realized that certain plants growing on seleniferous soils were capable of taking up selenium (Se) to very high levels. It is apparent that these plants, many of which are deeply rooted, may have a role in transporting Se from deeply situated deposits to the soil surface. Microorganisms associated with roots of these plants probably are important in many phases of plant up take of Se. It is known that Se is microbially oxidized from relatively insoluble forms (selenide and elemental Se) to mobile forms (selenite and selenate), microbial reduction of these products back to immobile forms is possible even in the presence of oxygen, and microorganisms are agents in volitilization of Se. Astragalus bisulcatus (Two-grooved milkvetch) and Astragalus pectinatus (Tine-leafed milkvetch) are common Se accumulating legumes on upland rangelands in Wyoming. Saprophytic microorganisms in the rhizospheres of these plants were shown to be relatively tolerant of selenite concentrations up to 30 ug g^{-1}. Further, the majority of the organisms in these rhizospheres were capable of reducing selenite to elemental Se and probably to volatile Se compounds. Both plants were heavily infected with vesicular arbuscular mycorrhizal fungi under field conditions. In a greenhouse study, these fungi were able to infect A. pectinatus at 10 ug Se (as selenite) per gram of soil.

INTRODUCTION

Selenium is a trace element required in human and animal nutrition at dietary levels below 1 ug/g (Schwartz, 1965; National academy of science, 1980; Watson, 1985). However, at levels less than 10 ug/g, it becomes a systemic metabolic toxin (Oldfield, 1987). In most arid as well as certain non-arid areas of the world, elevated levels of Se are present in soils, plants and animals (Trelease and Beath, 1949; National Research Council, 1986). In arid soils, selenium occasionally occurs as elemental Se(0), sometimes as selenides(-2), but particularly as selenites (+4) and selenates (+6) (Bainbridge and Wegryzn, 1988; Berrow and Ure, 1989). Elemental Se and selenides (specifically metal selenides) are not very toxic mostly because of their low solubility. Selenites and selenates are soluble and toxic. Selenium accumulator plants will concentrate this element to levels sometimes in excess of 10,000 ug g^{-1} (Inhat, 1989), although 1 to 10% of this is more common. When consumed by livestock, plants with high levels of selenium have been reported to elicit acute and chronic intoxications (James et al. 1989). Plants noted as Se accumulators (e.g. A. bisulcatus and A. pectinatus) may take up selenium from deeply situated deposits. Decay of above ground portions of these plants release soluble inorganic and organic selenium in surface soils.

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[1] Department of Plant, Soil and Insect Sciences, P. O. Box 3354, Univeristy of Wyoming, Laramie, Wyoming. 82071

Microbial metabolism of Se was recognized early in this century (Lipman and Waksman, 1923; Levine, 1925). The capacity to reduce selenite has been used as a diagnostic criteria for a variety of microorganism including such pathogens as Salmonella sp. (McCready et al., 1966). However, the processes of oxidation, reduction and volitilization of selenium were not knit into a cycle until early in the 1970s. Shrift eluded to a selenium cycle as early as 1973. Volitilization of Se from soils was reported in 1968 by Abu-Erreish, et al., but the role of microorganisms was not suspected until the early 1970s. Apparently the earliest account of such a possibility was obscure work reported by Holzinger, 1974, although a myraid of more recent work has been published (e.g. Reames and Zoller, 1980; Thompson-Eagle et al., 1989, 1991). The salient points are that soil microorganisms, including saprophytes, actively transform Se compound from soluble, oxidized forms to less soluble, reduced forms; and that Se is volitilized by soil microorganisms, usually as a low molecular weight organic form.

Vesicular-arbuscular mycorrhizal (VAM) fungi are common inhabitants of roots systems of most terrestrial plants. Much work has shown that VAM fungi are important in plant phosphate nutrition (e.g., Gray and Gerdemann, 1969). Rhodes and Gerdemann (1978) demonstrated that these fungi can be instrumental in enhancing uptake of sulfur compounds. They demonstrated that VAM infected Allium sp. accumulated 12 times as much labeled sulfur as did uninfected plants. Sulfur is a close analog to selenium in terms of its chemical and physical properties (McNeal and Balistieri, 1989). Rogers and Williams (1986) have shown that VAM fungi mediate plant accumulation of some non-nutrients such as cesium. In this study, it was shown that roots of VAM infected Melilotus officinalis accumulated twice as much Cs as did uninfected roots. The upshot of this work is that VAM fung may well be instrumental in the uptake of Se by plants, including those plants noted for their high Se accumulation habit.

The purpose of this research was to investigate how rhizosphere organisms of A. bisulcatus and A. pectinatus influence accumulation of selenium by these plants. Central hypothesis: soil microorganisms are involved in chemical transformations of selenium which may increase or decrease selenium availability to plants. The objectives were to (1.) examine saprophytic microorganisms in the rhizosphere of A. bisulcatus for the capacity to transform Se, and (2.) ascertain the role of mycorrhizal fungi in Se accumulation.

MATERIALS AND METHODS

FIELD SITES: The Laramie basin of SE. Wyoming has numerous seleniferous parent materials and consequently numerous seleniferous soils. Two seleniferous sites were located in Albany County: (1.) the

Tucker property (1.5 km west of Laramie) and the Harris Ranch (2.5 km west of Bosler). Soils excavated to 1 meter showed AB-DTPA extractable Se ranging from 0 to 176 ng g^{-1} of selenium. The most common Se accumulating plant on these sites was A. bisulcatus although A. pectinatus was also present. These plants typically bioaccumulate Se to levels at least 1000 time the soil extractable level.

RHIZOSPHERE SAPROPHYTIC ORGANISMS: Soils were taken from the root zone (rhizosphere) of A. bisulcatus at the Tucker property, serially diluted into sterile water, and 10^{-3} through 10^{-6} dilutions replicate plated on a solid mineral salts medium [(Williams and Wollum, 1981): 6.10 g K_2HPO_4, 2.02 g KH_2PO_4, 0.2 g $MgSO_4 \cdot 7H_2O$, 0.006 g $CaCl_2 \cdot 2H_2O$, 0.003 g $FeSO_4 \cdot 7H_2O$, 2.24 g $(NH_4)_2SO_4$, 10 g glucose, 15 g agar and distilled water to bring total volume to 1 liter (constituents were separated into three groups prior to mixing, each was steam-sterilized separately to minimize interactions. Groups were: (i) Glucose, (ii) Mg, Fe, Ca, N, and agar, and (iii) Phosphates)] containing 0, 1, 2, 4, 8, 16 or 30 ug g^{-1} of Se as selenite. Organisms were counted at 2, 3, 5, and 7 days and proportion of selenium reducing organisms determined. Selenium reducing organisms are distinctive because colonies develop a red color as selenite is reduced to selenium metal.

VESICULAR ARBUSCULAR MYCORRHIZAL FUNGI: As plants were collected at field sites, roots were separated and stained for V. A. mycorrhizal fungi (acid fuchsin-lactic acid method of Kormanik and McGraw, 1982). In a greenhouse study, A. pectinatus was germinated and grown in native soil (Vertic Torriorthent) from the Harris ranch which had been supplemented with elemental selenium or selenite selenium to 10 ug g^{-1}. The purpose to this preliminary study was to ascertain the tolerance of V.A. fungi to Se materials.

RESULTS AND DISCUSSION

RHIZOSPHERE SAPROPHYTIC ORGANISMS: The majority of saprophytic microorganisms from the rhizosphere of A. bisulcatus at the Tucker place were capable of reducing selenite to elemental selenium. Further, the microorganisms showed some intolerance to elevated levels of selenite since there was some depression of total organism numbers at Se (as selenite) between 1 and 30 ug/G^{-1}. It was also apparent that the microorganisms examined here were fully capable of releasing selenium in volatile forms. Volatile Se compounds (H_2Se, CH_3SeH and CH_3SeCH_3) have a distinctive odor (described as rotting garlic by some). Organisms growing on media containing selenite gave off this distinctive garlic odor during growth.

V. A. MYCORRHIZAL FUNGI: VAM fungi were demonstrated to be abundant on the fine roots of both A. bisulcatus and A. pectinatus. Our findings that Se accumulating plants are infected by V.A. mycorrhizae, leads to the hypothesis that VAM fungi are involved in the accumulation of Se compounds by some plants. The preliminary greenhouse study certainly demonstrated the ability of VAM fungi which infect A. pectinatus to tolerate high soil selenium.

SUMMARY

1. Astragalus bisulcatus and Astragalus pectinatus accumulate selenium to levels of roughly 1000 times that of extractable soil Se.

2. Saprophytic rhizosphere microorganisms from A. bisulcatus were capable of reducing selenite to elemental Se and further to volatile Se compounds.

3. Rhizosphere microorganisms were highly tolerant of media levels of selenite up to 30 ug g-1 media.

4. A. bisulcatus and A. pectinatus are infected by V.A. mycorrhizal fungi in the field. V.A. fungal infection on A. pectinatus was not deterred by soil levels of added selenite up to 10 ug Se g^{-1} soil.

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REVEGETATION SUCCESS AND BOND RELEASE: WHERE DO WE STAND?

Victoria J. Bryan[1] and Russell F. Price[2]

ABSTRACT

To a surface coal mining operator, the primary indicator of a successful operation is the efficient and profitable mining of coal. To a regulatory authority, the primary indicator of success is reclamation that is performed in accordance with the permit and the regulations following mining. To satisfy the reclamation goals of the statutes and regulations administered by States and the Federal Office of Surface Mining Reclamation and Enforcement (OSM), all parties involved must strive to achieve a nexus among a number of regulatory elements. These include the bond release regulations, environmental performance standards, revegetation success standards, and the approved reclamation plan designed to return the mined land to a specific use. The process of mining and reclamation is dynamic and greatly affected by economic and geographic factors. The challenge, then, is to administer permits, mine coal, and reclaim land in response to ever changing conditions and to strive to avoid conflicts among the elements that are used to determine successful reclamation.

This is not an easy task. Reclamation plans must be well designed, detailed enough that they can be used as a "scope of work" similar to a construction project, and must remain current. Reclamation plans must specify methods for complying with environmental performance standards and be designed to meet the ultimate tests of revegetation success. The standards for reclamation success are not fully formulated in all cases. With respect to revegetation, the regulatory requirements that appear to be the most complex and challenging include: revegetation success standards, normal husbandry practices, and methods to measure success. Some States have included approved success standards in their regulatory programs. However, most regulatory authorities are in the process of developing success standards and methods to measure reclamation success. To assure timely bond release, mining companies and regulatory authorities need to work together to develop detailed, specific reclamation plans and success standards for achieving and measuring reclamation success. Steps necessary for successful reclamation and timely bond release include following approved reclamation plans, monitoring results, and implementing remedial actions when needed. Waiting until the bond release application is submitted to address these issues is too late.

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[1] Victoria J. Bryan, Reclamation Bonding Specialist, Western Support Center, Office of Surface Mining Reclamation and Enforcement.

[2] Russell F. Price, Chief, Technical Assistance Division, Western Support Center, Office of Surface Mining Reclamation and Enforcement.

INTRODUCTION

The purpose of the Surface Mining Control and Reclamation Act of 1977 (P.L. 95-87; 30 U.S.C. 1201 et seq) (SMCRA) is to assure the nation an adequate supply of coal while at the same time, assure that the environment and agricultural productivity of the land are protected. The Office of Surface Mining Reclamation and Enforcement (OSM) regulates coal mining and administers SMCRA in States that do not have an approved State Program. These include the States of California, Tennessee, and Washington. OSM also regulates coal mining on Indian lands. Primacy States that have approved State Programs regulate coal mining in their States based on the requirements of SMCRA. In these cases, OSM has oversight responsibilities for the various State Programs.

A major requirement of SMCRA is for mined land to be reclaimed contemporaneously with mining. A reclamation plan written by the permit applicant and approved by the regulatory authority is the basis for the reclamation work. Developing reclamation plans requires comprehensive planning and forethought. Many of the mines under OSM's jurisdiction are large operations consisting of thousands of acres and are located in arid, semi-arid, and humid climates. Before a bond may be released in arid climates, the permittee's responsibility period (liability period) for revegetation is 10 years following the last augmented seeding, fertilizing, irrigating, or other work that would not be considered a long-term or "normal" land management practice. The responsibility period for revegetation in humid areas is 5 years following the last augmented work performed on the site. Reclamation plans are centered around a specific postmining land use appropriate for the area. The postmining land uses may include grazing, pastureland, cropland, forestry, recreation, fish and wildlife habitat, and industrial or commercial sites.

In order to obtain release of the performance bond, reclamation work must be performed in accordance with a variety of standards and regulatory requirements. If reclamation plans are not specifically designed to incorporate these requirements, or if on-the-ground work does not follow the approved plan, then release of the performance bond will be delayed.

Another major factor that may delay bond release is any uncertainty about the standards that will be used to judge whether the reclamation work is satisfactory. SMCRA requires that the standards used to measure success be included in the approved permit and in the approved regulatory program. Consequently, OSM and many State regulatory authorities are in the process of defining these standards and deciding what statistically valid methods will be used to measure success. This work is of the utmost importance. Uncertainty about the reclamation success standards that will be used can penalize the industry and delay release of all or part of the reclamation bond. Regulatory authorities, industry, and academia all need to better define what they expect from reclamation and be able to apply appropriate methods to judge success when reclamation is completed. Proactive planning throughout the permitting process, mining, and reclamation operations is necessary for gaining timely bond release.

Revegetation success alone does not mean that a reclaimed mine site is eligible for final bond release. However, revegetation is a complex operation involving numerous variables that are evaluated on a quantitative basis. If revegetation does not meet the standards for success, bond release will be denied. Because revegetation success is paramount to obtaining bond release, this paper focuses on requirements within the Federal regulatory framework for revegetation success. The requirements within the Federal regulations are applicable to approved State Programs that include similar requirements. Some of the difficulties experienced by regulatory authorities in defining and applying revegetation success standards are discussed.

REGULATORY REQUIREMENTS

To satisfy the reclamation goals of SMCRA, permittees must comply with a number of regulatory elements. The regulatory elements that apply to revegetation include: 1) the bond release regulations at 30 CFR 800.40; 2) the environmental protection performance standards at 30 CFR 816.111, .113 and .114; and, 3) the revegetation standards for success at 30 CFR 816.116.

The first set of regulations at 30 CFR 800.40 establish a timeframe for bond release based on specific stages or phases of work completed. For example, a phase I release of up to 60 percent of the bond amount may be granted after the mined area has been backfilled, regraded (which may include replacing topsoil), and the drainage system reestablished according to the reclamation plan.

The second phase of release relates to the revegetation stage of reclamation. If an area has been revegetated according to plan, vegetation has been established and the area is not contributing more than the allowable level of suspended solids to streamflow or runoff outside of the permit boundary, then OSM may release an additional amount of the bond. If the reclaimed area is prime farmland, the bond is not eligible for a release under this second phase of reclamation until productivity equals premining yields for 3 crop years in accordance with 30 CFR 823.15(b)(3).

As part of both phase I and phase II bond release actions, OSM must recalculate the cost to OSM if OSM had to perform remaining reclamation before deciding how much of the bond may be released. In all cases, bond sufficient to cover the cost of reestablishing the vegetation must remain in place until the final bond release occurs.

After all requirements of the reclamation plan have been met and the period of responsibility for revegetation has expired, OSM may approve a final, phase III bond release. The period of responsibility for revegetation is based on annual rainfall. Mines located in areas receiving more than 26 inches annually have a 5 year responsibility (liability) period. Mines located in areas receiving less than 26 inches annually have a 10 year period of responsibility (liability) as outlined at 30 CFR 816.116. The responsibility period begins after the last augmentation to the revegetation. Work considered to be augmentative is any seeding, fertilizing and/or irrigating not documented and approved by OSM as a "normal husbandry practice." The decision on what constitutes a normal husbandry practice is based on the land management practices considered to be normal for the area where the mine is located for land uses simiiar to the postmining land use approved in the permit. Any work done to the revegetated area that, if discontinued, would cause the revegetation to fail or not meet the success standards would be considered augmentative rather than a normal husbandry practice. To avoid continuous extensions of the responsibility period and long delays in receiving bond release, permittees under Federal jurisdiction need to work with OSM to gain a clear understanding of what work can be done that will not re-start the responsibility period.

The second set of regulatory requirements that need to be addressed in the reclamation plan are the general environmental protection performance standards for revegetation at 30 CFR 816.111, .113, and .114. Numerous requirements exist to assure that the revegetated area can support the approved postmining land use. These include seeding and planting to provide a diverse, effective, permanent vegetative cover that is comprised of native or introduced species necessary to accomplish the target postmining land use. In addition, the revegetation must be able to control erosion, have the same seasonal characteristics of the original vegetation, be capable of plant succession and self-regeneration, be compatible with plant and animal species of the area, and meet regulatory requirements with respect to poisonous and noxious plants and introduced species.

The third set of regulatory requirements pertaining to revegetation are the standards for success. As outlined at 30 CFR 816.116, these requirements include meeting all of the general performance standards discussed above, plus meeting criteria for ground cover, production, or stocking that are representative of unmined land in the area. With respect to measuring revegetation for success, reclamation is considered successful if the measurements for cover, production, and stocking are not less than 90 percent of the success standard established in the approved permit. These regulations require OSM to use a 90 percent statistical confidence interval (i.e., one-sided test with a 0.10 alpha error). In addition, the regulations at 30 CFR 816.111 require OSM to select standards for success and statistically valid sampling techniques and include them in approved Federal programs for the areas under its jurisdiction. Similar requirements are imposed on States with primacy that regulate coal on Federal lands within their boundaries.

Within the regulations pertaining to bond release, two distinct levels of revegetation "success" are included. The first of these is at 30 CFR 800.40(c)(2) where the phrases "successful revegetation has been established" and "revegetation that has been established" appear. In this context, "successful revegetation" has been interpreted to mean that revegetation must be established according to the reclamation plan and demonstrate enough growth to control erosion (Federal Register, 1991). Further, to be eligible for a phase II bond release, the revegetation must meet the general requirements at 30 CFR 816.111 pertaining to diversity, species composition, self-regeneration and plant succession. Requirements for cover and production do not have to be met at phase II unless the postmining land use is prime farmland (Federal Register, 1991).

The second reference to "success" is at 30 CFR 800.40(c) (3), which in turn refers to 30 CFR 816.116. These are the success standards pertaining to revegetation and include cover, production, and stocking as well as the requirements of 30 CFR 816.111. In this context, success for final bond release means that the reclaimed area must meet the standards for success established in the permit at the confidence level required for all the vegetation parameters evaluated.

RECLAMATION PLANS

Coal mining companies are required to develop reclamation plans that demonstrate how the performance standards of the regulations at 30 CFR Part 816 will be met. Reclamation plans must also include the proposed methods to be used to measure whether the success standards have been met. If the success standards are not clearly defined by the regulatory program and in the reclamation plan, then how will OSM determine that a bond release is warranted?

An analogy can be drawn between a coal mining operation and a construction project. Mining requires coordination between the mining operation and the reclamation operation that follows mining. On the mining side of the project, the contract (or mining plan) is based on meeting specifications for delivery of so many tons of coal to a customer. On the reclamation side of the project, the clients are the general public, surface land owners, and land management agencies who benefit by being able to use the reclaimed land. The reclamation plan section of the permit is the contract written to satisfy the performance standards of the regulations that are like design specifications under a construction contract. Filing a performance bond (like filing a contractor's completion bond) is a condition of the contract (mining permit) and release of the bond can only be done when the contract requirements (success standards) have been met. Local forestry, wildlife, agricultural, and land management agencies play the role of "consultants" to the mining project. The ultimate decision on whether the project was completed according to the design specifications and codes (success standards) rests with the regulatory authority which serves a function similar to a building/construction inspector.

If the contract requirements (success standards) are not specific, and if the contract (reclamation plan) does not clearly specify how the contractor will construct the project to meet the criteria, on what will the building inspector (regulatory authority) base its decision? If it is not possible to determine that a building has been built to code because the requirements are vague or subjectively applied, the project could be held "in limbo" indefinitely. Meanwhile, the contractor (mining company) would have money and equipment tied up in the project, and the financial backer (surety or bank) would have the performance guarantee obligated and unavailable for other projects. In addition, the clients (public and landowners) would not be able to occupy and use the property. In a scenario like this, everyone loses. In some cases, this describes a current dilemma faced by OSM, State regulatory authorities, and the coal mining companies they regulate. Efforts are currently underway to resolve these issues and prevent delays in releasing bond by developing success standards and statistically valid sampling techniques to evaluate revegetation.

REVEGETATION SUCCESS STANDARDS

In 1991, State regulatory authorities and OSM participated in a major joint effort sponsored by OSM to exchange information and to identify problems and possible solutions for defining success standards and sampling techniques (Forum, 1991). Measuring the success of reclamation is complex and requires developing standards that provide a balance between utility to the approved postmining land use, sound ecology, and site-specific conditions. Participants in the discussions on revegetation described their current use of specific success standards and voiced concerns about how to deal with standards not yet developed.

Under the Federal program, the frame of reference OSM uses to measure success in meeting the standards varies with different postmining land use designations. For example, when measuring ground cover and production for grazingland and pastureland, the revegetation is either compared with cover and production on a designated reference area or compared with technical standards approved by OSM in the approved mining and reclamation permit. Likewise with cropland, crop production of the reclaimed area is compared with crop production on a reference area or compared with an approved standard. If the postmining land use is fish and wildlife, recreation, or forestry, revegetation success is based upon tree and shrub stocking and vegetative ground cover. Stocking rates are developed in consultation with appropriate land management agencies and are based on local and regional conditions.

As mentioned above, many regulatory authorities are in the process of developing success standards for all vegetation parameters for all areas under their jurisdiction. As discussed at the bond release forum (Forum, 1991), regulatory authorities employ many different approaches to evaluating revegetation.

A number of States located in arid climates in the West reported successful use of reference areas to evaluate revegetation. However, States located in other geographic regions said that reference areas do not work. For example, when applicable to the postmining land use, two Western States indicated they use standards based on comparisons with undisturbed native ground. These States think that reference areas are relatively easy to manage, and that it is possible to locate range in good condition (Soil Conservation Service Ratings) for use as reference areas.

On the other hand, a Midwestern State with a humid climate reported that the use of reference areas does not work well because in areas with 40 inches of annual precipitation, native sites are not stable and plant succession is continuous. This State also said that the permit areas are too small to implement the effective use of reference areas.

A State located in the arid Southwest stated that reference areas are not an effective basis for annual comparisons since range conditions are poor and the range is abused in many areas. Also, due to geomorphic variability and the resulting diversity in vegetation, reference areas are not well suited for use as a comparison with postmining topography that is flat and rolling.

One State located in an arid climate discussed its use of historical vegetation data collected over a period of 3 or more years from an applicable reference area as a means of establishing standards. On the other hand, a Midwestern State discussed its use of standards based upon the condition and characteristics of the soils in the premined area, taking into consideration the postmining vegetation production level expected by the surface landowner.

A State in the West discussed its method for establishing technical standards. Here, data on vegetation by morphological types is collected over a 6 year period. Parameters measured and later used to develop standards include vegetation cover, density, and diversity. This State requires that 51 percent of the postmining species be native species. Reference areas are maintained until the post mining revegetation meets the success standard.

In geographic areas where the use of reference areas is not an effective way for developing standards to measure all vegetation parameters for success, it is clear that technical success standards must be developed. Academia could provide a great service to industry and the regulatory community alike by assisting in the development of technical standards for the various post mining land uses. It is essential to develop standards that meet the intent of the regulations, provide for practical, efficient reclamation, and allow timely bond release.

The extent of work left to do with respect to developing success standards and sampling techniques is considerable. Currently, 7 of the 24 primacy States have regulatory programs with approved standards and sampling techniques for part of the vegetation parameters and for husbandry practices. Three of these seven States have standards for all of the vegetation parameters. Four other States have regulatory programs that include approved success standards but not approved sampling techniques. With respect to the four Federal program States and the Indian Lands programs, one Federal program includes revegetation success standards but not sampling techniques. When considering vegetation parameters on an individual basis, the percentage of regulatory programs that include standards for each vegetation parameter for each land use is relatively low. For example, of the 24 primacy State programs, the 4 Federal programs, and the Indian Lands programs, 44 percent have a standard for cover; 31 percent for production; 34 percent for stocking; 17 percent for diversity; 13 percent for seasonality; 20 percent for erosion control; 17 percent for regeneration; 17 percent for permanence; and 24 percent have defined approved husbandry practices (Revegetation Task Force). Given that many regulatory programs currently lack standards and sampling methods, it is important for approved permits to include success standards, sampling methods, and husbandry practice requirements that will be applied when evaluating revegetation success for bond release.

To illustrate how bond release can be delayed or denied because of unclear technical standards in a permit and/or the lack of technical standards in a regulatory program, a hypothetical reclamation operation is discussed below. In this example scenario, the permit was issued with a post mining land use of forestry and wildlife. In the reclamation plan, the permittee stated that fertilization would occur after the initial seeding took place and that additional applications would be made as needed during the growing seasons.

General revegetation guidelines developed by the regulatory authority stated that for wildlife areas, the revegetation must consist of a high percentage of native species. Therefore, the approved seed mix included native species; but the required percentage of native species was not specified in either the approved permit or in the success standards of the regulatory program.

The reclamation plan specified a technical success standard for ground cover based on data collected over a period of years from a reference area using the point intercept method. During the final years of the responsibility period, the permittee used a different method to evaluate the ground cover on the reclaimed area. The change in methods meant that the sampling technique approved in the permit had not been used, and the data generated from the two techniques were not comparable.

Following reclamation and the end of the extended responsibility period, the permittee applied for bond release. After the area was inspected for bond release, the permittee was notified that the bond release could not be approved for several reasons: 1) the permittee employed a management practice (fertilization) during the extended responsibility period that the regulatory authority considered to be augmentative; 2) the percentage of native species present in the permanent revegetation was too low; and 3) the ground cover data could not be used to evaluate whether ground cover requirements had been met. The permittee thought that the approved reclamation plan provided the flexibility to decide when to apply fertilizer since the plan allowed for applications on an "as needed" basis. This ambiguity in the reclamation plan and the lack of approved husbandry practices in the regulatory program led to a major difference of opinion on this issue.

After the bond release was denied and the responsibility period restarted by the regulatory authority, the permittee requested a hearing. The regulatory authority's decision to deny bond release was upheld. Denying the bond release could have been avoided.

Had the regulatory authority included approved husbandry practices in its regulatory program, the permittee would have known what practices would restart the responsibility period. The permit could have been specific about when fertilizer could be applied, and what operations constituted approved husbandry practices. With respect to cover and diversity requirements, the regulatory program and the permit could have been specific. The permittee could have followed the approved permit with respect to the sampling technique and/or applied to revise the permit. The results of ongoing reclamation could have been monitored to assure that the revegetation was on target with the permit. This would have allowed problems to be identified earlier and a permit revision applied for if warranted.

To assure that, in reality, a scenario such as this does not occur, permittees and regulatory authorities must work together to establish the standards against which revegetation will be compared.

CONCLUSIONS

Revegetation success is a major factor in determining whether an area is eligible for bond release. Currently not all regulatory programs and not all permits include specific standards for judging revegetation success. While some States do have success standards in their approved programs, and efforts are being made to establish approved sampling techniques, more work needs to be done.

Where standards have not yet been developed, the regulatory community, with help from industry and academia, must work to develop standards. The standards must be realistic and practical enough to provide for compliance with reclamation requirements and timely bond release for the permittee. Eventually, these standards need to be included in approved State and Federal programs. To further enhance timely bond release, the regulatory authority must monitor compliance with the approved reclamation plan on a continuous basis to evaluate progress toward meeting the success standards. Regular inspections of revegetation are also needed to identify and resolve problems, all with the goal of meeting the success standards. If on-the-ground revegetation and sampling techniques demonstrate results that differ from what is needed to meet the success standards, some remedial action must be taken. One option is for the company to alter its reclamation methods to get back on target with the success standards. If the problem is not the method nor the practice but rather standards that are not clear or not possible to meet, the company could apply to revise the reclamation plan. Waiting until the final bond release inspection to find out that the success standards have not or cannot be met, even though a reclaimed area "appears" to be successful, is a mistake that can cost a company thousands of dollars and years of time. As part of this effort to monitor and evaluate results, the regulatory authority also has a responsibility to assure that its program and the permits it approves contain clear, concise, and well founded success standards and sampling techniques to encourage good reclamation and allow timely bond release. Successful reclamation and bond release allow the previously mined land to be returned to other uses, satisfying a major goal of SMCRA.

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

ABANDONED BENTONITE MINE SPOIL RECLAMATION-12 YEARS OF RESEARCH

G.E. Schuman[1], E.J. DePuit[2],
E.M. Taylor, Jr.[1] and J.L. Meining[3]

ABSTRACT

Reclamation of abandoned bentonite spoils is extremely difficult because of the high clay content, high salinity, high sodicity, absence of topsoil, and the semiarid climate of the region. Field research established in 1979 using sawmill wastes, gypsum, and nitrogen fertilizer as spoil amendments demonstrated that these amendments were effective in modifying the spoil characteristics and enabled successful revegetation. This research led to the development of a reclamation technology for abandoned bentonite spoils in 1985, which has been used to revegetate approximately 6500 hectares of abandoned bentonite spoils and has the potential to be used to reclaim an additional 15,000-20,000 hectares of spoils in the region.

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[1] High Plains Grasslands Research Stn., USDA-ARS, 8408 Hildreth Rd., Cheyenne, Wyoming.

[2] Dept. of Natural Resource Sciences, Washington State University, Pullman, Washington.

[3] Wyoming Dept. of Environmental Quality, Land Quality Division, Cheyenne, Wyoming.

INTRODUCTION

Reclamation of abandoned bentonite mine spoils is extremely difficult because of the high clay content, high salinity, high sodicity, absence of topsoil, and the semiarid climate of the Northern Plains Region. Field research established in 1979 (Schuman and Sedbrook 1984) using sawmill wastes (bark, chips, and sawdust) and nitrogen fertilizer as spoil amendments demonstrated that the amendments enabled water infiltration into the spoil and the establishment of a perennial grass/shrub community. Research demonstrated that 40 tonnes/hectare of sawmill wastes and 5.0 kg N/tonne of wood residue resulted in good spoil-water relations, good seedling establishment and subsequent forage production (Smith 1984, Smith et al. 1985). Further study of the revegetated ecosystem showed that the wood residue amendment and subsequent plant establishment also enabled the leaching of soluble salts from the surface 45 cm. However, because of the large pool of soluble and exchangeable sodium the exchangeable-sodium-percentage (ESP) increased, indicating a potential for deterioration of the system (Belden 1987, Belden et al. 1990). Therefore, gypsum was surface applied to the revegetated spoil at the rate of 56 tonnes/ha in the spring of 1987. This rate of gypsum was adequate to reduce the ESP from 48% (pretreatment level) to the desired 15%. The gypsum elicited a positive response within the first year after treatment and within three years had reduced the ESP of the surface 15 cm by over 32%, with significant improvement in the ESP over the entire 60 cm profile (Meining 1991, Schuman and Meining in press). Gypsum amendment also significantly increased the spoil-water storage, demonstrating an improvement in the physical characteristics of the spoil (Schuman and Meining in press). These short-term benefits indicate the treatments should produce continued improvement in the sodicity of the revegetated spoil and ensure sustainability of the system. Application of gypsum also resulted in significant increases in the aboveground biomass of the revegetated plant community (Moore et al. 1991). Decomposition of the sawmill wastes indicate that microbial decomposers are functional. Decomposition was found to increase significantly with increased nitrogen fertilizer rates. Maximum decomposition after five years was approximately 28% and did not increase significantly with nitrogen applications above the 5.0 kg N/tonne of wood residue fertilizer rate (Schuman and Belden 1991). This research led to the development of a reclamation technology for abandoned bentonite mine spoils, which has been used to revegetate approximately 6500 ha of abandoned bentonite mine spoils and has the potential to be used to reclaim an several thousand additional hectares of abandoned bentonite spoils in the region (Richmond 1991).

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Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

THE CITY OF ANACONDA EROSION CONTROL AND STABILIZATION OF "C" HILL

Larry Holzworth[1], Jerry Schaefer[2], Glen Green[3], and Tim Wiersum[4]

ABSTRACT

Prior to the late 1800's, the "C" hill was vegetated by a mosaic of Douglas-fir and aspen intermixed with grassland. The hill was excessively logged to supply fuel for the adjacent copper smelter. The deforestation, heavy metal and sulfide pollution from the smelter stack fallout as well as steep slopes, and a harsh climate inhibited plant growth and resulted in severe soil erosion. Sediments were deposited onto the streets and into storm drains.

Increased street maintenance costs in the city of Anaconda, as well as the concerns of noxious weed invasion and the aesthetics of the area, resulted in the development of The City of Anaconda Erosion Control and Stabilization of "C" Hill project in 1986.

Surface smoothness was manipulated with a Modified Hodder Gouger™, which excavated minibasins to ameliorate acid soil surfaces and to trap water for plant establishment and growth. Lime was surface applied, following basin construction, according to soil test results. A native grass seed mixture and fertilizer were broadcast over the pits, and straw mulch was spread over part of the seeded area.

Shrubs and trees were planted to provide long-term stabilization to the hill. Tree and shrub seeds were collected from adjacent native species and grown under contract for planting on some of the disturbed sites following 2 years of weed control.

Good stands of grass, shrubs, and trees have been successfully established on "C" hill and have provided erosion control. The total cost of the project has been offset by decreased maintenance costs, restored productivity, and aesthetic benefits to the city of Anaconda.

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[1] Plant Materials Specialist, Soil Conservation Service (SCS), Bozeman, MT 59715.

[2] Economist, SCS, Bozeman, MT 59715.

[3] District Conservationist, SCS, Deer Lodge, MT 59722.

[4] Soil Conservationist, SCS, Missoula, MT 59801.

INTRODUCTION

Prior to the late 1800's, the foothills surrounding Anaconda, Montana--locally referred to as the "A" and "C" hills--were vegetated by a mosaic of Douglas-fir and aspen stands intermixed with grasslands. Copper refining began in the Anaconda area in the late 1800's and continued until 1980. Initially, the smelters were located north and across the valley from the project area. In the early 1900's, the smelter was moved adjacent to and east of the project area. The smelter site, tailing piles, and sediment ponds east and downslope from the study area have been declared an EPA Superfund cleanup site. In the late 1800's and early 1900's, the forested hills adjacent to the smelter were extensively logged. Immediately after the logging operations, much of the foothill area exhibited major soil erosion problems. Removal of the natural vegetative cover, combined with heavy metal and sulfide smelter pollution of the soil, as well as a harsh climate, has inhibited the reestablishment of plant cover (USDA-Soil Conservation Service, 1986).

Adams et al. (1986) reported only 20 percent of the project area vegetated prior to treatment. Plant cover and diversity increased with distance from the smelter. The most impacted areas are within a 5-mile radius of the smelter, inline with the prevailing wind. Studies show these sites have lost up to 16 inches (40.6 cm) of topsoil. Infertile, poorly structured subsoils--sometimes referred to as desert pavement--caused by past surface erosion comprise the upper part of the soil profile. Runoff from the project area has caused flood damage and sediment deposition in downtown Anaconda. Increased maintenance costs, concern for noxious weed invasion, and visual quality of the area resulted in the development of this cooperative erosion control project.

MATERIALS AND METHODS

The 664-acre (268-ha) project area is located in southwestern Montana near the city of Anaconda. The elevation ranges from 5,500 feet (1,676 m) near town to 7,193 feet (2,192 m) on the "C" hill south of town. The mean annual precipitation (MAP) is 14 inches (356 mm), with June being the wettest month. The growing season ranges from 100 to 117 days. The "C" hill has a predominantly northern exposure with an average 25 percent slope.

The project area was divided into three treatment units based on similar site characteristics. Only the work on a 51-acre (20.6-ha) area will be reported in this paper. This area was divided into two sites--a 12-acre (4.9-ha) east location, and a 39-acre (15.8-ha) west area. These sites were selected because they could be transversed with reclamation equipment. They were treated in 1987 and 1989, respectively. The areas contain a mixture of two main geologic types: quartzites and extrusive, fine-grained, volcanic rocks.

The soils are eroded and heavily polluted with copper, iron, lead, zinc, cadmium, and arsenic in the surface 2 to 3 inches (5.1 to 7.6 cm) of soil. The heavy metals have become phytotoxic due to the acidifying effects of the sulfide stack emission fallout. The erosion rates are 24 tons-per-acre (53.8 t ha^{-1}) per year. The allowable soil loss for these soils is 3 tons-per-acre (6.7 t ha^{-1}) per year (USDA-Soil Conservation

Service, 1978). (Erosion greater than the allowable tolerance means that the soil is eroding faster than it is being formed.)

The area has 15 to 35 percent slopes, very gravelly loam surface textures, and a very gravelly clay loam subsoil derived from volcanic and quartzite rock. The surface 2 to 3 inches (5.1 to 7.6 cm) varies from 4.2 to 5.3 pH. The pH range improves to 6.2 and 7.7 at the 2- to 10-inch (5- to 25.4-cm) depth (Long, 1986). The area consists of a mixture of rock fragments, erosion pavement, bare soil, patches of grass, clumps of quaking aspen *Populus tremuloides* Michx., and an occasional Rocky Mountain juniper *Juniperus scopulorum* Sarg.

The reclamation strategy was to scrape away patches of the acid soil surface 2 to 3 inches (5.2 to 6.7 cm), and plant suitable indigenous species into the basic pH subsoil (Adams et al., 1986, and Schafer, 1988). Minibasins were constructed with a Modified Hodder Gouger™[1/] (USDA-Forest Service, 1977) on the east and west sites during April 1987 and April 1989, respectively. The excavated basins were approximately 3 feet long by 1 foot wide (.91 x .3 m), and 4 inches (.1 m) deep, with alternating centers at 1 foot (.3 m) in between basin rows. The basins were orientated lengthwise downslope. The final surface modification resulted in approximately 7,260 basins per acre (17,932 per ha).

Fertilizer and ground lime were broadcast over the basins following the seeding. Lime was applied with an all-terrain spreader at a rate of 2.2 tons-per-acre (4,931.7 kg ha^{-1}) (Western Fertilizer Handbook, 1985). This consisted of an agricultural grade lime passing through an 80-mesh screen and testing 85 percent $CaCO3$ equivalent. Two hundred pounds-per-acre (224.2 kg ha^{-1}) of 16-20-0 NPK was applied with the Modified Hodder Gouger™ during broadcast seeding.

Adams et al. (1986) collected soils from the "C" hill at various depths and conducted bioassay studies at Montana State University using six plant cultivars. The following species' recommendations were ascertained from their study.

A native species cultivar mixture containing, by weight, 50 percent 'Magnar' basin wildrye *Leymus cinereus* (Scribner & Merrill) A. Love, 40 percent 'Critana' thickspike wheatgrass *Elymus lanceolatus* (Scribner & J. G. Smith) Gould, and 10 percent yellow blossom sweetclover *Melilotus officinalis* (L.) Lam. was prescribed for the west site. Approximately 23 pounds-per-acre (25.8 kg ha^{-1}) pure live seed were broadcast with the Modified Hodder Gouger™ into the minibasins in April 1987. On the east site, native species cultivar mixture, by weight, was 40 percent 'Magnar' basin wildrye, 30 percent 'Critana' thickspike wheatgrass, and 30 percent 'Sodar' streambank wheatgrass *Elymus lanceolatus* (Scribner & J. G. Smith) Gould. Approximately 24 pounds-per-acre (26.9 kg ha^{-1}) pure live seed were broadcast with the Modified Hodder Gouger™ into the minibasins in April 1989. One-half of the west side planting was mulched with 3,000 pounds-per-acre (3,362.5 kg ha^{-1}) of barley straw.

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[1] Trade names used solely to provide specific information. Mention of a trade name does not constitute a guarantee or endorsement by the U.S. Department of Agriculture, Soil Conservation Service.

The cost of establishing grass was $873 per acre. (See table 1 for an itemization of expenses.) This amount does not include the $65 per acre that was spent on a one-time, weed control application.

Table 1. Per Acre Cost of Grass Seeding

Grass	$203
Pitting and Seeding	55
Fertilizer	32
Lime	286
Lime Application	55
Straw Mulch	242
Total Cost Per Acre	$873

Shrub and tree seeds were collected from native stands adjacent to the reclamation site during 1986 and 1987. Species collected were Rocky Mountain maple *Acer spicatum* Lam., serviceberry *Amelanchier alnifolia* Nutt., rose *Rosa woodsii* Lindl., buffaloberry *Shepherdia argentea* Nutt., dogwood *Cornus stolonifera* Michx., aspen *Populus tremuloides* Michx., and oregongrape *Berberis repens* Lindl. Seedlings were propagated by the Montana Department of State Lands, Division of Forestry; State Forest Tree Nursery; and Bitterroot Growers (a private nursery). From 1989 through 1992--following grass establishment and weed control--21,800, 1-year-old, 20-cubic-inch (327.7-cm^3) "cone-tainer" nursery stock seedlings were transplanted into the grass seedings by city of Anaconda personnel. One, .35-oz (10-gm), Agriform 20-10-5 fertilizer tablet was planted with each seedling to provide a 2-year nutrient source.

In 1989, seedlings were planted in the upper 10 acres (4 ha) of the west area. There were 360 seedlings per acre (865 plants per ha) planted at a spacing of 11 by 11 feet (3 x 3 m). In April 1990, the remaining 25 acres (10 ha) were planted with the same seedling species and numbers per acre. The east area treatment site was planted in April 1991 with the same seedling species, but at a rate of 680 plants per acre (1,630 plants per ha) on an 8- by 8-foot spacing (2.44 x 2.44 m). The cost of planting shrubs was $1,006 per acre (table 2).

Table 2. Per Acre Cost of Shrub Planting

Plants ($.70 per plant x 680 plants)	$476
Planting Plants ($.70 per plant)	476
Fertilizer ($.08 per plant)	54
Total Cost Per Acre	$1,006

Percent ground cover was estimated in 20 constructed minibasins, using the line-intercept transect method. Data was collected during September 1988 for initial cover within the 1987 planting, and in September 1992 for established cover. Cover was estimated during September 1992 within the 1989 planting. Also during September 1992, three, random, 9.6-square-foot (.89-m^2) plots on the west site and five on the east site were harvested for total herbage yield. The percent cover and herbage yield from the treatment areas were compared to bare ground prior to reclamation. Tree and shrub survival was monitored by the city of Anaconda for each planting year. MAP recorded for the city of Anaconda was used as an estimate for the reclamation sites.

RESULTS AND DISCUSSION

Grass Seeding Evaluations

West Area 1987 Planting

The initial percent ground cover transect was estimated in November 1987, after the first growing season. Percent cover (includes standing live and dead plant material and surface residue) in the basins ranged from 1 percent up to 30 percent (figure 1). Plant vigor was generally low, with plants in only a few basins showing good vigor. This site was heavily infested with Canada thistle *Cirsium arvense* L., which affected the grass seedling vigor and stand establishment.

Thickspike wheatgrass was the dominant species in 65 percent of the basins; with basin wildrye present in 95 percent, but dominant in only 15 percent. The yellow sweetclover was present in only 25 percent and showed low vigor and growth.

Plant establishment in interspaces in-between the basins was observed in 65 percent of the transect plots.

The 1992 data collected in the 20 original basins along the permanent transect showed dramatic improvement in percent cover and plant vigor (figure 1). A large part of the improvement was due to chemical control of the Canada thistle on the site. Percent ground cover ranged from a low of 40 percent (which occurred in the single basin with a low of 1 percent in 1987) up to 100 percent in one-half of the basins.

Basin wildrye became the dominant species in 50 percent of the pits. The yellow sweetclover is no longer present in any of the basins along the transect. One native grass species that is becoming more dominant in plant composition is redtop *Agrostis alba* L. This plant is the dominate species occurring in the undisturbed areas between the basins.

The straw mulch was blown off the windward portion of the treatment area and heaped onto the downwind sites. This excessive mulch cover was detrimental to seedling establishment.

Vegetative clippings for grass production were taken in October 1992. No previous data for comparison is available. Three plots were clipped, using a 9.60-square-foot (.89-m) frame, and produced an average air-dry herbage yield of 563 pounds-per-acre (631 kg ha^{-1}) within and in-between the basins (figure 2).

East Area Planted 1989

Two transects were set up on this area for determining percent ground cover. No previous data was collected from this area for comparison, but a comparison can be made with the area of gravel pavement and no vegetative cover before the reclamation treatment. A total of 20 basins along each transect were evaluated.

Cover ranged from no cover in one basin to 100 percent in four basins (figure 1). Overall, plant vigor was very high in all the basins.

Chemical control of spotted knapweed *Centaurea maculosa* Lam. and Canada thistle resulted in the very high plant vigor of the seeded species.

Streambank wheatgrass was the dominant grass species in 82 percent of the basins evaluated, with basin wildrye being the dominant species in only 15 percent. Thickspike wheatgrass was never a dominant component and was absent in 32 percent of the basins. Average air-dry herbage yield was 340 pounds-per-acre (381-kg ha^{-1}) during 1992 (figure 2).

Interbasin plants, comprised primarily of streambank wheatgrass, occurred in-between 70 percent of the pits. Unlike the west area treatment, very little redtop was present in this area.

Tree and Shrub Plantings

The objective of the tree and shrub planting was to have a minimum of 100 plants per acre established within 10 years after planting. They could provide a minimum of 25 percent cover, which would reduce the current erosion by 50 percent. Survival rate of the seedlings ranges from 70 to 95 percent.

CONCLUSION

It remains to be seen how effective the treatment measures will be in achieving the long-term goals of the project; however, there have been some very immediate and observable benefits resulting from the treatment measures. The grass cover catches and holds more of the winter snow, and the basins allow more of the rainfall and snowmelt to infiltrate the soil profile, rather than becoming runoff. This allows favorable moisture regimes for maintaining established vegetation, and encourages vegetation to spread and establish between the basins.

Noxious weeds are still a problem and will have to be treated annually to maintain the health of the grass stands and insure high tree and shrub seedling survival.

The area is stable and aesthetically pleasing with the grass cover, and will improve as the trees and shrubs mature. Community pride in the project has increased as seen in less litter and vehicle damage to the area. Less runoff is occurring from the slopes, which means less sediment coming onto the city streets and into the storm drains, reducing city maintenance costs.

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Figure 1. Total percent ground cover.

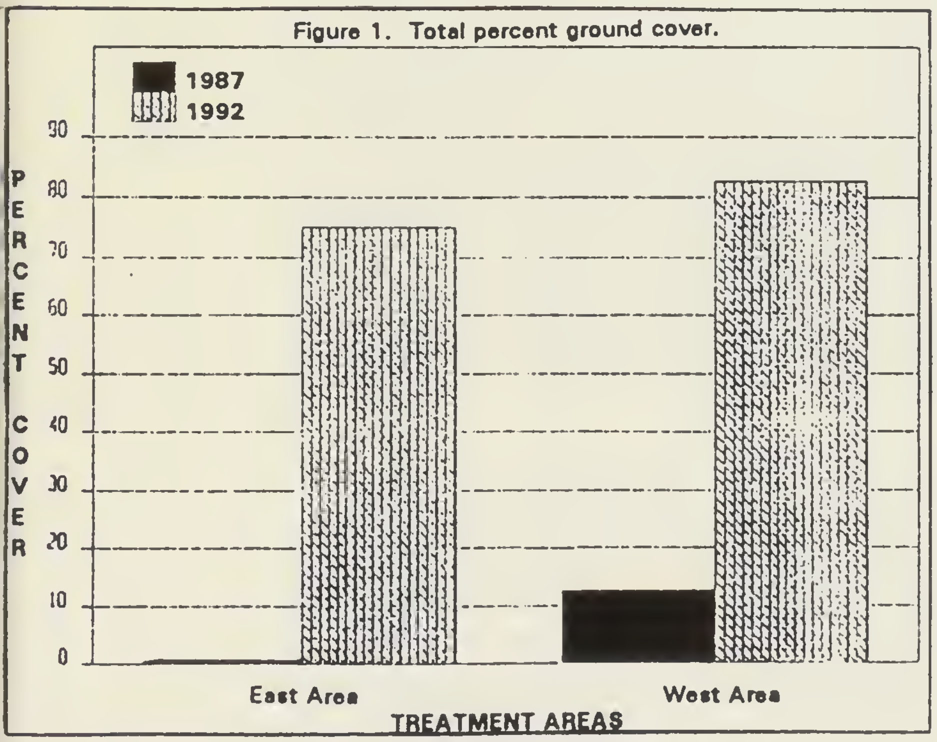
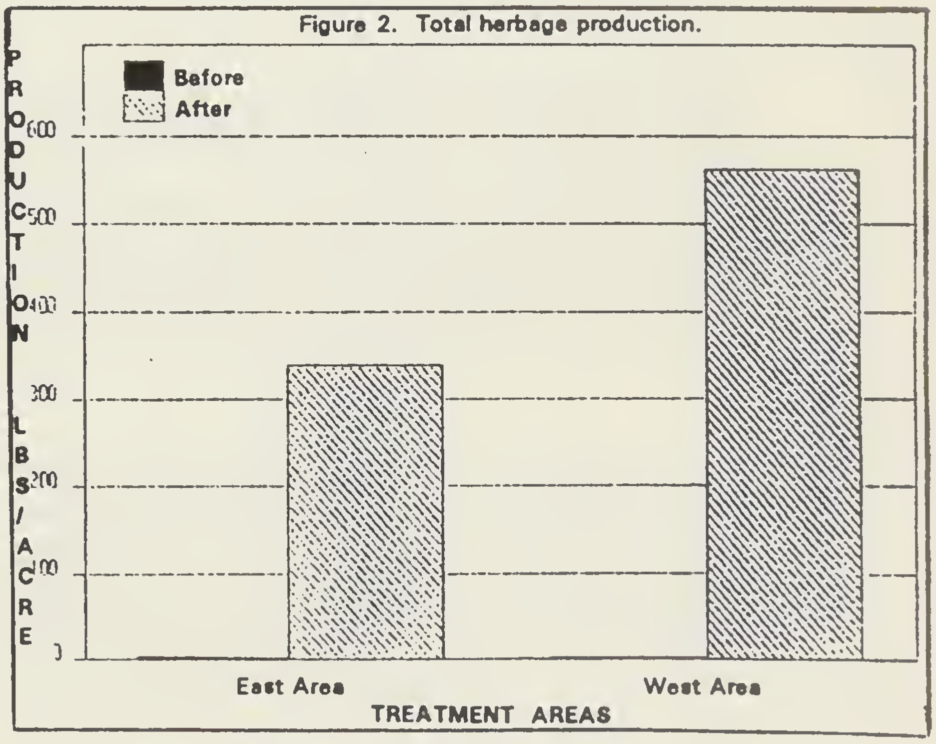


Figure 2. Total herbage production.



Planning, Rehabilitation and Treatment of Disturbed Lands
Billings Symposium, 1993

USING ORGANIC MATTER TO SELECT OPTIMUM TOPSOIL SALVAGE DEPTHS: A STRATEGY TO IMPROVE TOPSOIL QUALITY AND REDUCE SALVAGE DEPTHS

K.D. Coppinger[1], P.R. Ogle[1], and D. McGirr[2]

ABSTRACT

Topsoil salvage and replacement is an important component of successful reclamation programs for surface coal mines. However, because state and federal laws often require salvage of all suitable topsoil, many mines are salvaging and stockpiling volumes of topsoil that exceed the amount necessary for adequate reclamation. Because soil fertility tends to diminish with depth, deeper soil salvage may result in the dilution of more fertile surficial soils with poorer quality subsurface soils when replacement occurs. This may result in an overall reduction in the quality of soils used for reclamation. Therefore, the potential exists to improve the quality of topsoil being used for reclamation by reducing salvage depths.

The objective of this study was to develop a method to evaluate optimum topsoil salvage depths and apply it in a site-specific situation. Because soil organic matter (OM) directly or indirectly controls numerous environmental factors that affect soil fertility and productivity in the Powder River Basin (e.g., soil moisture, nutrient availability, microbial activity, soil structure, and oxygen availability), OM content was the principal factor used in the soil evaluation method.

The procedure to evaluate optimum salvage depths included: 1) establishment of the maximum stripping depth for the soils of interest based on physical and chemical soil suitability criteria currently used by Wyoming Department of Environmental Quality (WDEQ); 2) establishment of a standard for minimum OM content of soils to be salvaged; 3) development of soil OM profiles to determine depths at which OM becomes less than the standard; 4) determination of what depth must be salvaged to meet topsoil volume requirements; and 5) development of a plan to selectively salvage soils with the highest OM content, unless topsoil volume requirements necessitate salvage of soils with lower OM levels.

This method was applied to soils within The Carter Mining Company's Caballo Mine permit area, Powder River Basin, NE Wyoming, where surface soils (0-24 inches) depth generally have greater than 0.5% OM (the standard set for this study) and subsurface soils often have less than 0.5% OM. The analysis resulted in a 43% reduction in the volume of topsoil to be salvaged and stockpiled. The final replacement depth slightly exceeded the 18 inches required by WDEQ, providing a necessary reserve of topsoil in case of unforeseen soil shortages.

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[1] Mariah Associates, Inc., 605 Skyline Drive, Laramie, Wyoming.
[2] The Carter Mining Company, P.O. Box 3007, Gillette, Wyoming.

INTRODUCTION

Topsoil salvage and replacement are important reclamation techniques for surface coal mine operations in the eastern Powder River Basin of Wyoming because topsoil usually provides the best available plant growth medium on reclaimed areas (Barth and Martin 1982, Cook et al. 1974, U.S. Forest Service [USFS] 1979). Topsoil, due to the presence of plant propagules, higher organic matter, and microorganisms, is generally a superior plant growth medium to spoil (Ogle and Rendente 1988). Because it is a valuable resource, Wyoming Department of Environmental Quality, Land Quality Division (WDEQ-LQD) *Rules and Regulations* (1989) require mines to plan for salvage, storage, and replacement of all suitable soils that will be disturbed by mining.

Soil suitability is determined using WDEQ-LQD criteria which include assessment of soil pH, electrical conductivity, saturation percentage, texture, sodium absorption ratio, selenium and boron contents, and the volume of coarse fragments (WDEQ-LQD 1984). This method is defined as the physical and chemical (PC) method in this paper. At times greater than 60 inches of soils may be suitable under the PC criteria, thus many mines are handling and stockpiling volumes of topsoil that exceed the amount necessary for adequate reclamation.

The objective of this study was to develop a method to evaluate optimum topsoil salvage depths. A procedure to identify and salvage only those soils that are most suitable for reclamation was designed. The procedure also considers salvage depths that are necessary to meet topsoil volume requirements.

METHOD

The method to evaluate optimum topsoil salvage depths was developed in four stages:

- evaluation of the effects of topsoil depth on plant establishment and productivity on reclaimed areas,
- identification of soils attributes that promote plant growth in the northern Great Plains,
- determination of the availability of selected attributes in soils to be disturbed, and
- analysis of topsoil volume requirements for reclamation.

Information obtained during these four phases was used to develop a step-by-step procedure that could be used by numerous industries to assist in planning topsoil salvage and replacement.

An evaluation of effects of topsoil depth on plant establishment and productivity on reclaimed areas and identification of soil attributes that promote plant growth were made using published results of reclamation and soil fertility research. Soils data and soil volume requirements from the Caballo Mine permit area were obtained from the baseline soils inventory for the mine permit (Carter Mining Co. 1978). Additional soils data were obtained from the Caballo Mine's annual prestripping soil testing results.

RESULTS AND DISCUSSION

Effects of Topsoil Depth on Reclamation Potential

Results of research conducted in the late 1970's and early 1980's show that salvage and replacement depths of 9 to 33 inches of topsoil promote the highest rates of plant establishment and greatest productivity on reclaimed areas in semi-arid environments (Barth and Martin 1982, Halvorson et al. 1986, Pinchak 1983, Schuman and Taylor 1978, USFS 1979). Optimum topsoil replacement depth is defined as the depth at which the rate of increase of plant establishment or productivity, as a function of topsoil depth, becomes small or static. The optimum replacement depth depends on numerous factors including, but not limited to, topsoil and overburden quality, the amount and distribution of precipitation, and topographic position.

Overburden quality has been shown to have a significant effect on plant establishment and productivity (Barth and Martin 1982). Over neutral overburden, topsoil depths of 9 to 24 inches promoted optimum establishment of perennial grasses. Optimum productivity was obtained with 11 to 33 inches of topsoil. Maximum productivity of a mixture of perennial grasses and forbs was obtained with 24 inches, and maximum diversity was achieved with 16 inches. Deeper soils were required to obtain optimum establishment and productivity over sodic and acid spoils.

Barth and Martin (1982) and Pinchak (1983) demonstrated that optimum topsoil replacement depth was also influenced by precipitation and topographic position. In years with normal precipitation, optimum productivity occurred with 16 to 20 inches of topsoil. Deeper soils augmented productivity in years with higher than average precipitation or in sites at lower topographic positions. In drier years, optimum productivity occurred in shallower soils.

Organic Matter as a Soil Quality Criterion

With the exception of sunlight and carbon dioxide, the environmental components essential for plant growth are strongly influenced by soil organic matter (OM) (Figure 1) (Smith et al. 1987, Whitford and Elkins 1986). As a source of carbohydrates, soil OM provides the raw materials for microbial metabolism. Microbial activity converts organic molecules, such as organic nitrogen, into plant-available nutrients, such as nitrate. Soil OM also affects soil structure and bulk density. Soils with higher OM contents have lower bulk densities, greater porosity, and greater infiltration potential (Pinchak 1983). Greater porosity and infiltration potential promote the storage and availability of soil water, which is often the

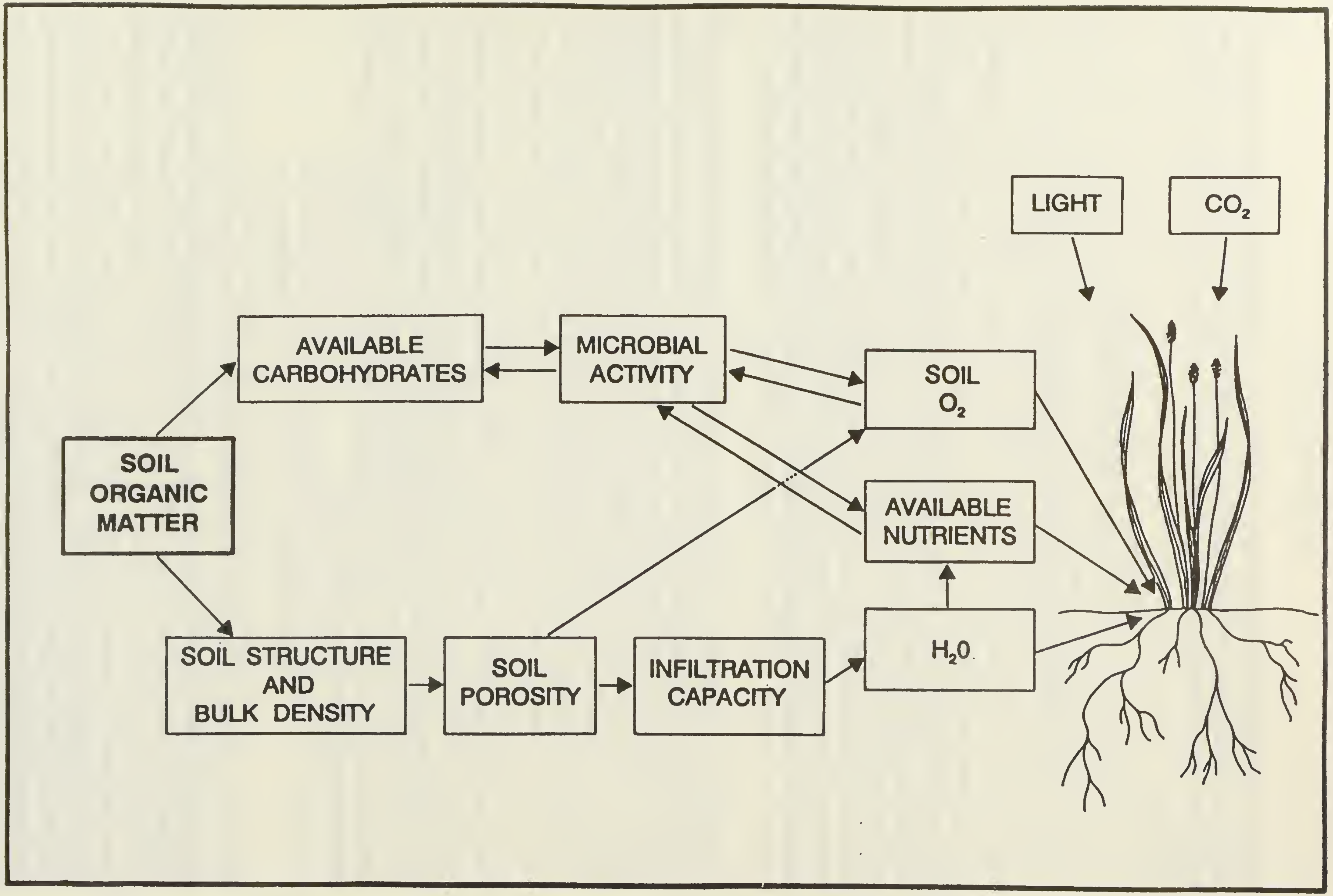


Figure 1 Effects of Soil Organic Matter on Components Essential for Plant Growth.

principal limiting factor to plant growth in semi-arid regions (Barth and Martin 1982, Cook et al. 1974, Monson and Plummer 1978, Pinchak 1983).

Because OM controls many of the environmental factors affecting plant growth, the percentage of OM in soils was selected as the method to evaluate soil quality as a function of depth. Only those soils classified as suitable for reclamation under the PC method were examined.

A Method to Evaluate Optimum Topsoil Salvage Depths Based on Soil Organic Matter Content and Volumetric Requirements

A method to evaluate optimum topsoil salvage depths based on soil OM content was developed (Figure 2). The proposed evaluation process will allow mines and other industries to improve the overall quality of material used for reclamation by increasing the availability of soil OM and other beneficial resources typically concentrated in surface horizons such as active microbial populations and a diverse seed pool. After determining suitability, soil OM profiles can be used to determine appropriate salvage depths based on soil quality--the minimum OM content (0.5% in this study) may be based on site-specific conditions. The topsoil evaluation process also includes a periodic review of the volume of topsoil available so that replacement depth standards may be attained.

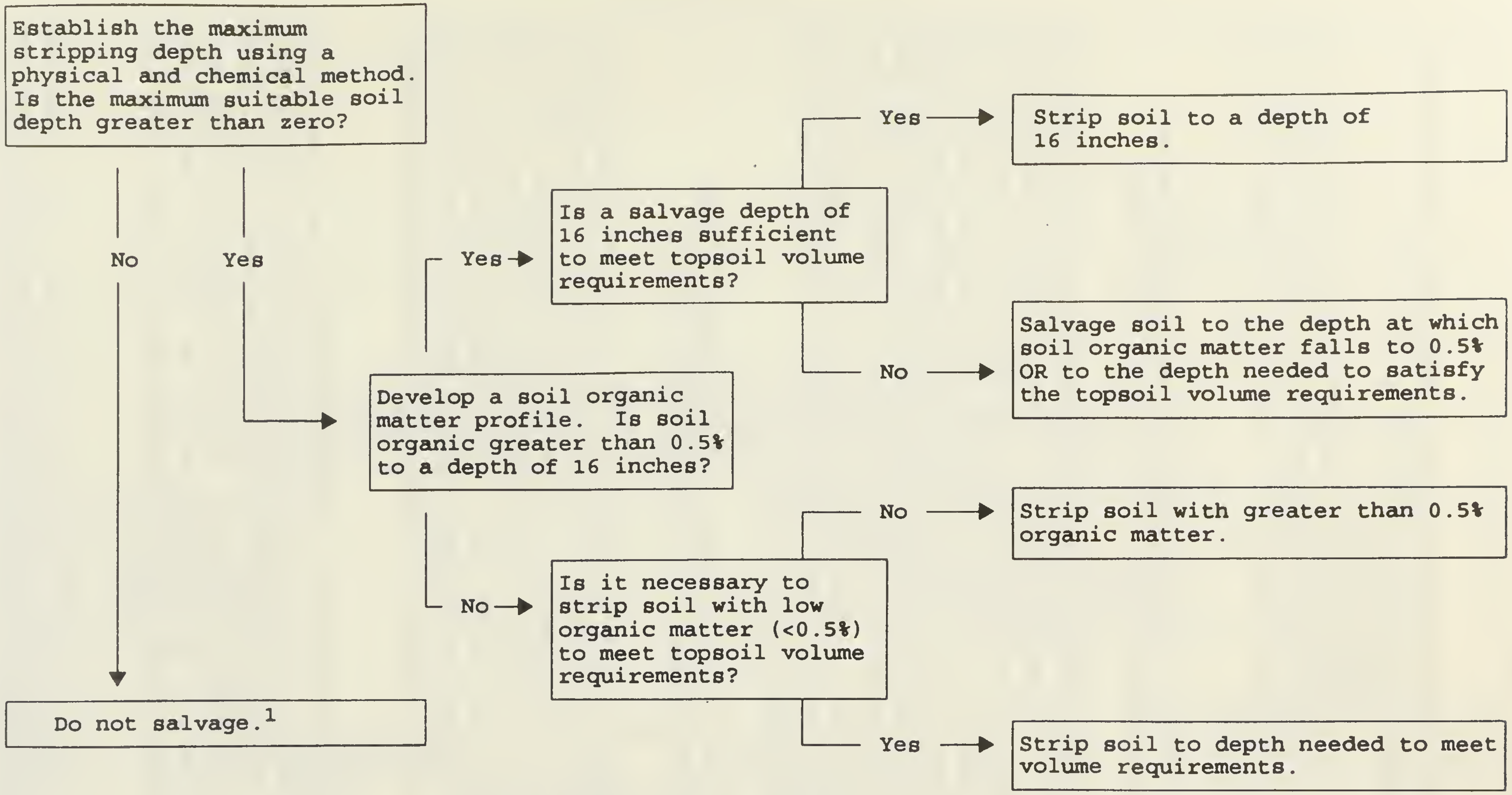
Application of the OM Method to Revise Salvage Depths For Soils Within the Caballo Mine Permit Area

Organic matter content in soils within the Caballo Mine permit area ranged from 0.1 to greater and 4.0%. In most soils, OM content decreases with increasing depth (Figure 3a); however, OM profiles may vary considerably (Figure 3b). Based on USFS (1979) criteria (> 1.5% OM = good suitability for reclamation, 0.5 - 1.5% OM = fair suitability, and <0.5% OM = poor suitability), most surface soils (0 - 24 inches) in the mine area have good to fair suitability for reclamation use. Using these criteria, many subsurface soils (below 24 inches) would be poor materials for reclamation.

Current salvage depths for these soils were determined during the permitting process for the Caballo Mine and were based on PC soil suitability criteria. Many soils in the study area have no limitations to great depths (e.g. Decolney soils, Figure 3c) and would be stripped to a depth of 60 inches. However, analysis of soil OM as a function of depth in many of these soils suggests that the quality of topsoil to be used during reclamation could be greatly improved by reducing salvage depths from current levels.

Salvage of only the upper portions of appropriate soil profiles would reduce the dilution of this material with deeper, poorer quality soils and would improve the overall quality of salvaged soils. Additionally, there would be less dilution of

ORGANIC MATTER METHOD


Establish the maximum stripping depth using a physical and chemical method. Is the maximum suitable soil depth greater than zero?
No
Yes
Do not salvage.[1]
Develop a soil organic matter profile. Is soil organic greater than 0.5% to a depth of 16 inches?
Yes
Is a salvage depth of 16 inches sufficient to meet topsoil volume requirements?
Yes
Strip soil to a depth of 16 inches.
No
Salvage soil to the depth at which soil organic matter falls to 0.5% OR to the depth needed to satisfy the topsoil volume requirements.
No
Is it necessary to strip soil with low organic matter (<0.5%) to meet topsoil volume requirements?
No
Strip soil with greater than 0.5% organic matter.
Yes
Strip soil to depth needed to meet volume requirements.

[1] If topsoil volume requirements necessitate limited salvage of unsuitable topsoil, a detailed plan for mitigating the adverse impacts of incorporating these soils into the topsoil stock on potential reclamation success should be developed, in cooperation with appropriate regulating agencies.

Figure 2 Decision-Making Process for Determining Optimum Topsoil Salvage Depths Using the Organic Matter Method.

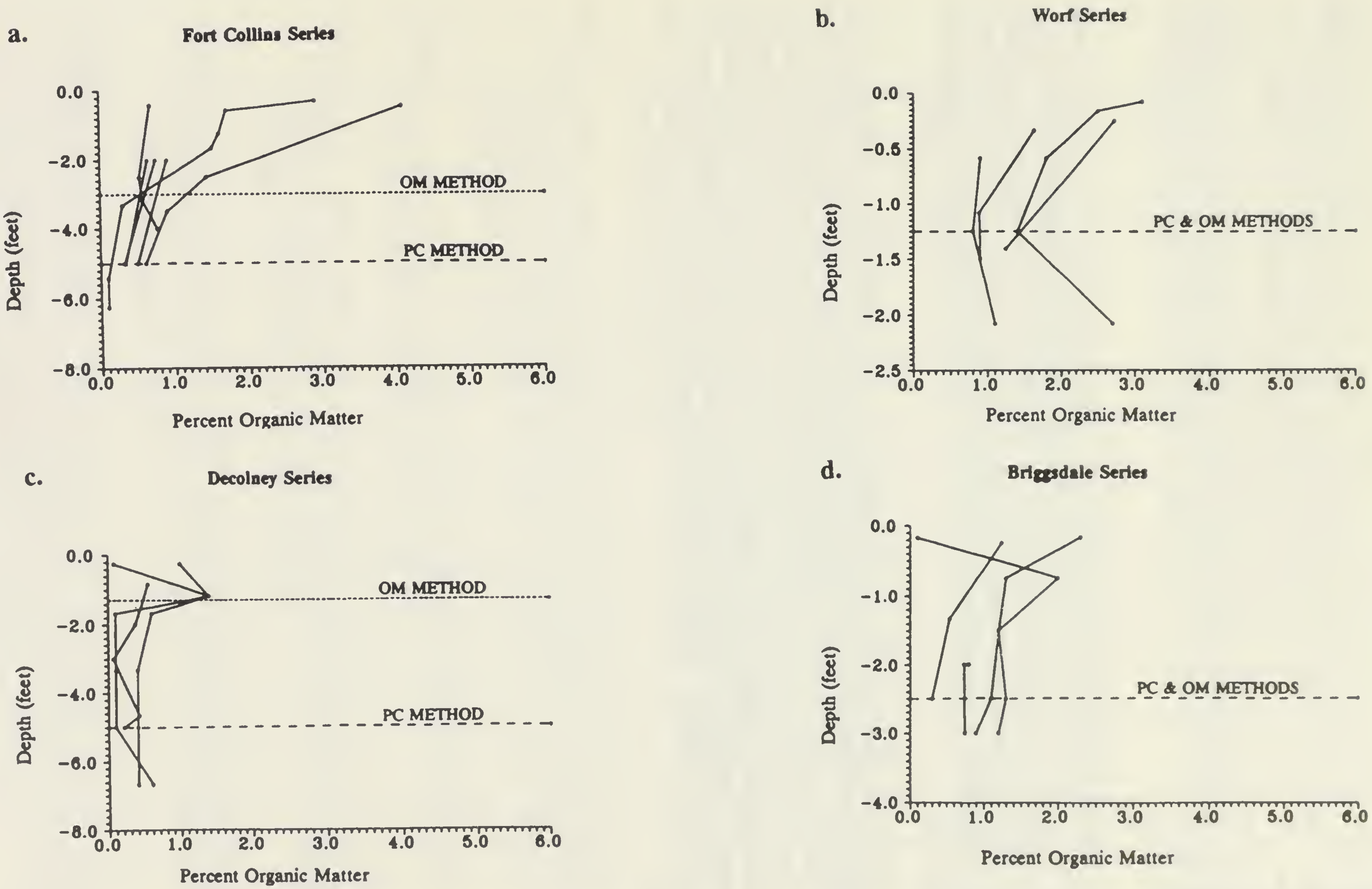


Figure 3 Soil Organic Matter Profiles for Selected Soils Within the Caballo Mine Permit Area, Campbell County, Wyoming.

microbial populations and viable seeds that are normally found in the upper portions of the soil profile (Miller et al. 1979, Beauchamp et al. 1975).

A graphical analysis of OM content as a function of depth was made for all soils in the Caballo Mine permit area for which soil OM data were available. Salvage depth for most soils were revised to 16 inches because research suggests this depth will promote establishment of diverse and productive perennial plant communities over neutral overburden (Barth and Martin 1984, Pinchak 1983). Coincidentally, soil OM levels are typically greater than 0.5% to this depth, suggesting that the bulk of biological activity in soils in the region would be concentrated in the surface 0 - 16 inches.

Salvage depths greater that 16 inches were recommended for selected soils with greater than 0.5% OM lower in the profile (Figure 3d). Assuming that the higher levels of OM are not due to the presence of relatively inert carbonaceous materials, greater salvage of these higher quality soils could be used to ensure that sufficient topsoil is available for reclamation, especially if any unsuitable soils or spoils are encountered.

The revised stripping depths recommended for the Caballo Mine resulted in an overall reduction of the volume of topsoil to be salvage, stockpiled, and replaced from 38,801 acre-feet to 21,931 acre-feet, a decrease of 43%. Overall replacement depth was reduced from 48 to 26 inches; still well above the 18 inch minimum replacement depth required by WDEQ-LQD for the Caballo Mine. Reduction of topsoil salvage depths within the Caballo Mine permit area will result in improved quality of soil material available for reclamation and reduced volumes of soil handled.

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Planning Rehabilitation and Treatment of Disturbed Lands
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THE ADAPTABILITY OF SELECTED TREE AND SHRUB SPECIES FOR SURFACE MINE RECLAMATION

Michael J. Knudson[1], Russell J. Haas[1], Dwight A. Tober[1], and Erling T. Jacobson[2]

ABSTRACT

Survival and growth rates of 16 deciduous and 4 coniferous tree and shrub species were evaluated at an abandoned surface coal mine in central North Dakota. Spoil ridges at the Wilton Mine were graded and contoured into three land treatments: level, wavelike, and tops level. These treatments simulated various degrees of surface mine reclamation. A fourth untreated area served as the check plot. Soil analysis indicated no major limitations at the site. The planting was established in 1976. Data was collected annually for six years, summarized, and a report prepared.

Ten years later, in 1991, data was again collected on survival and height. Of the 20 species planted, 11 of them had at least 25 percent survival. The table below shows the species survival by treatment. The best surviving species were deciduous shrubs. The only conifer which had less than 25 percent survival was the Black Hills white spruce. The only species to die on all plots was Carolina poplar. Overall survival was highest on the level plots and least on the wavelike plots. Results of the study indicate that grass and weed competition may be a more limiting factor than degree of land leveling and shaping.

Plant Survival in 1991

Species*	Level	Wavelike	Tops Leveled	Check	Total
caragana	7	8	9	7	31
golden currant	10	6	8	4	28
buffaloberry	10	3	10	5	28
chokecherry	10	5	6	7	28
common lilac	7	7	3	7	24
Siberian elm	8	4	6	2	20
Russian olive	5	2	9	1	17
ponderosa pine	6	2	4	4	16
Rocky Mt. juniper	1	4	4	6	15
eastern red cedar	0	5	4	6	15
green ash	4	1	4	6	15

*A total of 10 plants of each species were planted in each treatment type.

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[1] USDA Soil Conservation Service, Box 1458, Bismarck, North Dakota 58502
[2] USDA Soil Conservation Service, 100 Centennial Mall North, Lincoln, Nebraska 68505

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THE STATISTICAL DESIGN AND EVALUATION OF REVEGETATION DATA FOR DETERMINING REVEGETATION SUCCESS

Robert C. Postle

ABSTRACT

The Surface Mining Control and Reclamation Act of 1977 (SMCRA) requires the establishment of a diverse, effective, and permanent vegetative cover of the same seasonal variety native to the area and capable of self-regeneration and plant succession. Vegetative cover must be equal in extent to that of the native vegetation in the area. Final bond release cannot occur until these requirements are satisfied. The Federal regulations implementing SMCRA require the use of statistically valid sampling techniques for evaluating revegetation success. For ground cover, production, and stocking, the sampling techniques must use a 90 percent statistical confidence interval.

Based on the regulatory requirements, the use of statistics in evaluating revegetation success is mandatory. Statistics must be incorporated into the revegetation sampling program as well as the evaluation of the data for determining revegetation success. The statistical procedures that will be used to evaluate revegetation success should be defined in the permit application. Prior to sampling, existing data should be used to estimate the sample sizes to reach a 90 percent confidence interval for all parameters. Sampling schemes should be designed to ensure random location of sample points. As sampling is ongoing, sample adequacy should be recalculated using the new vegetation data. For production, this may involve the use of green weights in the field. Because of the variability in moisture content, final sample adequacy should be calculated using dry weights. Populations that are not always normally distributed, such as shrubs, may require very large sample sizes. Further stratification of the bond release or reference area based on homogeneous plant communities will reduce sample size. Transformation of the sample data is another option for normalizing the data. Comparisons of the vegetation data from the reclaimed area to the success standard, be it a reference area, historical record, or technical standard, can be made using a variety of statistical tests, including t-tests and analysis of variance.

Key Words: revegetation, sampling, statistics, sample adequacy

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[1] Ecologist, U.S. Office of Surface Mining Reclamation and Enforcement, 1020 15th Street, Brooks Towers, Denver, CO 80202.

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